



# THE UNIVERSITY *of* EDINBURGH

This thesis has been submitted in fulfilment of the requirements for a postgraduate degree (e.g. PhD, MPhil, DClinPsychol) at the University of Edinburgh. Please note the following terms and conditions of use:

This work is protected by copyright and other intellectual property rights, which are retained by the thesis author, unless otherwise stated.

A copy can be downloaded for personal non-commercial research or study, without prior permission or charge.

This thesis cannot be reproduced or quoted extensively from without first obtaining permission in writing from the author.

The content must not be changed in any way or sold commercially in any format or medium without the formal permission of the author.

When referring to this work, full bibliographic details including the author, title, awarding institution and date of the thesis must be given.

# Evaluating the use of marginal abatement cost curves applied to greenhouse gas abatement in the UK agriculture

*Vera Eory<sup>1,2</sup>*

*Doctor of Philosophy*

*The University of Edinburgh*

*2015*

<sup>1</sup> *School of Geosciences, University of Edinburgh*

<sup>2</sup> *Scotland's Rural College*



# Abstract

Climate change is arguably the most important global societal challenge. Developing ‘low-carbon societies’, i.e. reducing greenhouse gas (GHG) emissions and adapting to a changing climate, is becoming a policy goal across the globe. Agriculture plays an important role in this transformation. The sector is highly vulnerable to climate variability, and is a significant source of emissions. At the same time, it has potential for reducing GHG emissions and also provides opportunity for carbon sequestration in soils and crop biomass.

Policy support for mitigating GHG emissions is being informed by scientific evidence on the effectiveness and costs of mitigation opportunities. This information is frequently depicted in marginal abatement cost curves (MACCs), an assessment tool which can help to visualise the hierarchy of technical measures and their cumulative level of abatement. Similarly to other assessment tools, MACCs’ suitability to provide information has certain limitations. Furthermore, different derivations of MACCs are appropriate to answer different questions. In order to draw both informative and reliable conclusions for policy decisions, the characteristics of the MACCs and the resulting limitations have to be presented clearly.

This dissertation seeks to answer the general question whether the agricultural MACCs can be improved so that they provide more comprehensive and tailored information to policy makers. In particular five limitations of the MACCs are discussed: the lack of representation of wider effects, the issue of cost-effectiveness of policy instruments and the inclusion of transaction costs, the uncertainty in the MACCs, the boundaries and the heterogeneity of the analysis. Theoretical frameworks are developed and case study examples are provided for these limitations, and the frameworks are assessed in terms whether they achieve the goal of providing more comprehensive information to policy makers than a conventional MACC. Furthermore, the dissertation summarises the available methodologies and applications in agriculture to enhance the MACCs and provides guidelines for researchers and policy makers about the choice of methods and the communication of the results in order to improve the use of MACCs in the policy process.



# Declaration

I declare that this thesis and the papers within it have been composed by myself and that no part of this thesis has been submitted for any other degree or qualification. The work described is my own unless otherwise stated.

Vera Eory

*June 2015*



# Table of Content

Abstract.....	ii
Declaration.....	iv
Table of Content .....	vi
List of Tables .....	x
List of Figures.....	xi
Abbreviations.....	xii
Acknowledgments.....	xiv
1 Introduction.....	1
1.1 Greenhouse gas emissions from agriculture .....	2
1.2 Understanding MACCs .....	2
1.3 Addressing the limitations of the MACCs.....	5
1.3.1 Wider effects .....	8
1.3.2 Transaction costs and the cost-effectiveness of policy instruments .....	11
1.3.3 Uncertainty.....	12
1.3.4 Boundaries .....	13
1.3.5 Heterogeneity .....	16
1.3.6 Other limitations .....	16
1.4 Structure of the dissertation.....	20
2 Paper I. Co-benefits and trade-offs between greenhouse gas and air pollutant emissions for measures reducing ammonia emissions and implications for costing.....	24
2.1 Abstract.....	24
2.2 Introduction .....	25
2.3 Ammonia and greenhouse gas emissions in agriculture .....	27
2.3.1 Ammonia.....	27
2.3.2 Nitrous oxide.....	28
2.3.3 Methane.....	30
2.3.4 Carbon dioxide.....	31
2.4 Dietary options .....	31
2.5 Livestock housing.....	36
2.6 Manure storage .....	38
2.6.1 Liquid manure .....	38
2.6.2 Solid manure .....	40
2.7 Soil fertilisation .....	42



2.8 Conclusions .....	43
3 Paper II. Multiple-pollutant cost-effectiveness of greenhouse gas mitigation measures in the UK agriculture.....	46
3.1 Abstract.....	46
3.2 Introduction .....	47
3.3 Background.....	49
3.4 Methods .....	50
3.4.1 Calculations.....	50
3.4.2 Data sources .....	52
3.5 Results .....	58
3.5.1 Private and external cost comparison.....	58
3.5.2 GHG-MACC .....	60
3.5.3 MP-MACC-A and MP-MACC-B .....	60
3.5.4 MP-MACC-C and MP-MACC-D .....	62
3.5.5 MP-MACC-E .....	62
3.6 Discussion.....	65
3.7 Conclusion.....	68
4 Paper III. Evaluating the cost-effectiveness of agricultural mitigation policy.....	71
4.1 Abstract.....	71
4.2 Introduction .....	71
4.3 Agricultural policy and GHG mitigation .....	73
4.4 MACC analysis .....	75
4.5 Methods and data.....	76
4.6 Results .....	83
4.7 Discussion.....	84
5 Paper IV. Assessing uncertainty in the cost-effectiveness of agricultural greenhouse gas mitigation .....	89
5.1 Abstract.....	89
5.2 Introduction .....	89
5.3 Sources of uncertainty in the economic assessment of agricultural GHG mitigation.....	92
5.4 Uncertainty in the economic assessment of agricultural GHG mitigation.....	94
5.5 Data and Methods.....	98
5.6 Results .....	102
5.6.1 Uncertainty of the economically optimal GHG abatement .....	102
5.6.2 Uncertainty in the ranking of the mitigation measures .....	105
5.7 Discussion and conclusion.....	107
6 Paper V. Linking an economic and a life-cycle analysis biophysical model to support	

agricultural greenhouse gas mitigation policy .....	111
6.1 Abstract.....	111
6.2 Introduction .....	112
6.3 Methodology.....	113
6.3.1 Global Livestock Environmental Assessment Model .....	113
6.3.2 ScotFarm.....	115
6.3.3 Harmonising GLEAM and ScotFarm .....	115
6.3.4 Defining farm types .....	117
6.3.5 Case study: using sexed semen to reduce unwanted male calf numbers on Scottish dairy farms .....	118
6.4 Results .....	120
6.5 Discussion.....	122
7 Conclusions.....	125
References.....	133
Appendix – Papers published.....	155



# List of Tables

Table 1 Typical characteristics of the three MACC approaches.....	4
Table 2. Main limitations of the MACCs and examples for research targeting these issues...	6
Table 3. European N <sub>2</sub> O, CH <sub>4</sub> , CO <sub>2</sub> and NH <sub>3</sub> emissions and the contribution of main agricultural activities to the agricultural emissions (EU-27, 2011) .....	27
Table 4. Contribution of the different livestock species to CH <sub>4</sub> emissions from enteric fermentation and manure management (EU-28, 2011).....	31
Table 5. Matching measures between the two studies .....	52
Table 6. Unit damage cost sets used in this study.....	56
Table 7. Basis of valuation of damage costs.....	57
Table 8. Pollution savings from GHG measures with CE < SPC in 2022 in England and Wales expressed as % of BAU agricultural load .....	65
Table 9. Value of agricultural pollution load of GHG, NO <sub>3</sub> <sup>-</sup> , NH <sub>3</sub> , P and sediment in 2020 in England and Wales, £ million.....	66
Table 10. CE of MMs in the 2022 Scottish MACC and allocation to PIs .....	78
Table 11. Uptake, coverage and public policy operation cost assumptions for PIs.....	82
Table 12. CE and abatement potential PIs in 2022 .....	83
Table 13. An inventory of uncertainties in the economic assessment of agricultural GHG mitigation .....	95
Table 14. Mitigation measures analysed in this Chapter .....	98
Table 15. Characteristics of the three levels of uncertainty assigned to the input variables of the MACC model.....	100
Table 16. Lowest and highest value of the 95% CI of the mean of the economically optimal GHG abatement for different simulations.....	104
Table 17. Value of selected parameters for lactating cows.....	114
Table 18. Modelling differences between GLEAM and ScotFarm .....	117
Table 19. Typology of Scottish farms generated, based on FAS.....	118
Table 20. Differences between baseline and MM assumptions .....	119
Table 21. Main characteristics of the modelled baseline dairy farms .....	119
Table 22. Production, GHG emission and gross margin data (baseline and MM) .....	121
Table 23. Summary of the main limitations of the MACC with suggested approach when providing information to policy decisions .....	130

# List of Figures

Figure 1. Optimal pollution abatement .....	3
Figure 2. An illustrative MACC .....	5
Figure 3. Sources of GHG and NH <sub>3</sub> emissions on farm .....	26
Figure 4. ‘Hole-in-the-pipe’ model .....	29
Figure 5. Effect of inclusion of co-effects on the GHG MAC curve and on the economic optimum of pollution control .....	48
Figure 6. Annual private and social costs and the value of GHG savings. ....	59
Figure 7. MP-MACC-B .....	61
Figure 8. MP-MACC-E .....	63
Figure 9. CE of measures on the MACCs which have data on co-effects .....	64
Figure 10. Hypothetical engineering MACC (right) and the corresponding abatement potential deducted from an emissions baseline (left) .....	75
Figure 11. Abatement potential and overall CE of MM packages.....	84
Figure 12. Sources of uncertainty (in squares) in the climate change feedback loop .....	93
Figure 13. Effect of uncertainty on the optimal abatement level .....	97
Figure 14. The 95% CI of the mean of the economically optimal GHG abatement for the three parametric distributions: truncated normal, censored normal and triangular, for the three levels of uncertainty (narrow, medium and wide PDFs).....	103
Figure 15. The 95% CI of the mean of the economically optimal GHG abatement by propagation of uncertainty of individual uncertainty sources and all uncertainty sources combined, for three levels of uncertainty.....	105
Figure 16 The probability of the ranking of each MM .....	106
Figure 17. Uncertainty assessment of the input variables.....	108
Figure 18. Conceptual framework of the linkage between GLEAM and ScotFarm.....	116
Figure 19. Sensitivity of the CE to the price of the sexed semen and the EI of the suckler beef.....	121

# Abbreviations

AA	Amino acid
AD	Anaerobic digestion
BAU	Business as usual
C	Carbon
CAP	Common Agricultural Policy
CBA	Cost benefit analysis
CE	Cost-effectiveness
CEA	Cost-effectiveness analysis
CFP	Central feasible potential
CH <sub>4</sub>	Methane
CI	Confidence interval
CO <sub>2</sub>	Carbon dioxide
CO <sub>2</sub> e	Carbon dioxide equivalent
CP	Crude protein
DM	Dry matter
EI	Emission intensity
FAS	Farm Account Survey of Scotland
FFBC	Farming for a Better Climate
GHG	Greenhouse gas
GLEAM	Global Livestock Environmental Assessment Model
GWP	Global warming potential
HFP	High feasible potential
IPCC	Intergovernmental Panel on Climate Change
LCA	Life cycle analysis
LFP	Low feasible potential
MACC	Marginal abatement cost curve
MBC	Marginal benefit curve
MCA	Multi-criteria analysis
MP-MACC	Multiple-pollutant marginal abatement cost curve
MTP	Maximum technical potential
N	Nitrogen
N <sub>2</sub>	Di-nitrogen
NO	Nitric oxide
N <sub>2</sub> O	Nitrous oxide
NH <sub>3</sub>	Ammonia

$\text{NH}_4^+$	Ammonium
$\text{NO}_3^-$	Nitrate
$\text{NO}_x$	Mono-nitrogen oxides
$\text{N}_r$	Reactive nitrogen
P	Phosphorous
PI	Policy instrument
$\text{PM}_{2.5}$	Fine particulate matter
RDP	Rural Development Programme
RPP	Report on Proposals and Policies
SPC	Shadow price of carbon
SRDP	Scotland Rural Development Programme
TAN	Total ammoniacal nitrogen
VS	Volatile solids

# Acknowledgments



This research was undertaken within the Scottish Government Rural Affairs and the Environment Portfolio Strategic Research Programme 2011-2016 and with funding provided to ClimateXChange (<http://www.climateexchange.org.uk/>). For more information please see: <http://www.scotland.gov.uk/Topics/Research/About/EBAR/StrategicResearch/future-research-strategy/Themes/ThemesIntro>. Further funding was provided by the AnimalChange project which received funding from the European Community's Seventh Framework Programme (FP7/ 2007-2013) under the grant agreement n° 266018. The author is grateful to the UN FAO for permission to use GLEAM in this study.





# 1 Introduction

Climate change is one of the most pressing environmental problems we are facing today. As human activities are the major drivers behind global warming (IPCC 2013a), finding alternative ways for production and consumption is crucial in alleviating the harm climate change is likely to cause. However, transforming societies requires both individual and political will, both of them influenced by a range of factors, including the costs and the benefits of the transformation. The assessment of the impacts of alternative pathways on the environment and on the economy is therefore necessary for making informed decisions and designing efficient policies.

An assessment tool to analyse economically optimal greenhouse gas (GHG) abatement is the marginal abatement cost curve (MACC). MACCs have become rather popular in the past decades: they are being used to inform policy both about the estimated optimal level of mitigation effort and about the cost-effectiveness (CE) of possible mitigation measures (MMs). Examples include global, continent-level and country-level MACCs in different sectors of the economy – for an overview of the use of MACCs see Kesicki and Stratchan (2011). The MACCs' popularity is mostly due to its ability to convey information in a highly visual, relatively simple way. However, a number of limitations also exist. Addressing all of them at the same time could lead to a highly complex analysis, and difficulty in interpreting results. However, answers to specific policy questions would clearly benefit from addressing related limitations in conventional MACC analysis.

This dissertation seeks to answer the general question whether the agricultural MACCs can be improved so that they provide more comprehensive information to policy makers. In particular four limitations of the MACCs are discussed: the inclusion of wider effects in the MACC, the transaction costs and cost-effectiveness of policies, the uncertainties of the MACC and the question of boundaries. A theoretical framework is developed and a case study example is provided for each of these limitations, and these frameworks are assessed in terms of whether they achieve the goal of providing more comprehensive information to policy makers than a conventional MACC.

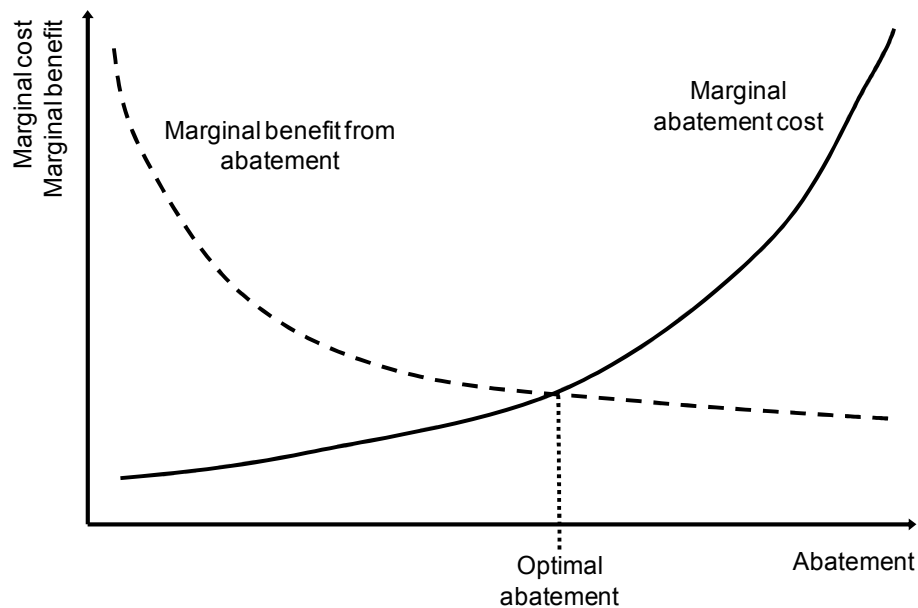
## 1.1 Greenhouse gas emissions from agriculture

Agricultural activities on farms have been estimated to account for approximately 12% of global, 10% of European and 9% of UK anthropogenic GHG emissions, not including the CO<sub>2</sub> (carbon dioxide) emissions and carbon sequestration effects of land use and land use change (European Environment Agency 2014, Smith *et al.* 2007, Thomas *et al.* 2011). These emissions are predominantly in the form of non-CO<sub>2</sub> GHGs: namely N<sub>2</sub>O (nitrous oxide) and CH<sub>4</sub> (methane). Most of the agricultural N<sub>2</sub>O emissions are produced in soils, with a lesser amount generated during manure management. The main sources are the nitrogen (N) added to soils (e.g. with inorganic and organic fertilisation, crop residues, atmospheric deposition, livestock excreta on pastures) and excreted by livestock in animal houses. Additionally, soluble nitrogen compounds leached into water bodies and gaseous ammonia (NH<sub>3</sub>) emissions can also be converted into N<sub>2</sub>O. Agricultural CH<sub>4</sub> emissions originate from the digestive system of animals, from manure stores, and from anoxic soils, like wetlands and rice paddies. In animals, methanogenesis happens during bacterial fermentation of feedstuff in the rumen of cattle, sheep and other ruminants, and also takes place, to a lesser extent, in the large intestine of all livestock. In manure management CH<sub>4</sub> is generated during the anaerobic decomposition of livestock bedding and manure, especially in liquid manure stores. CH<sub>4</sub> emissions from rice cultivation are globally important, but marginal in Europe.

## 1.2 Understanding MACCs

MACCs show the cost of reducing pollution by one additional unit as a function of the cumulative pollution reduction achieved against a business as usual (BAU) scenario (Figure 1). When compared to the marginal benefits from pollution reduction, the economic optimum of pollution reduction is defined as the intercept of these two curves (Pearce and Turner 1989). Up to the economic optimum the money spent on an additional unit of pollution reduction (e.g. carbon dioxide equivalent (CO<sub>2</sub>e)) would yield higher benefits from the avoided pollution, while beyond the economic optimum further spending on pollution reduction would yield less benefits than the money spent on abatement, indicating that the money could be spent better elsewhere. The marginal cost at the economic optimum suggests a pollution price or tax level which would theoretically allow achieving the optimal abatement. MACCs have spatial and temporal context, they refer to a country or specific

region, to the whole economy or to a sector and to a time period of either one year or more, often for a time period in the future.



*Figure 1. Optimal pollution abatement*

*Optimal pollution abatement is defined by the marginal cost of abatement and the marginal benefits from abatement (Pearce and Turner 1989)*

The marginal cost of abatement can be calculated in various ways. Vermont and De Cara (2010) group MACCs into three main types based on the methodology used to derive the curves. The first is based on micro-economic models, where the behaviour of the economic agents is modelled to derive the marginal cost of abatement, usually assuming profit-maximising agents, with the prices exogenously defined. An example for this approach is a spatial assessment of agricultural non-CO<sub>2</sub> mitigation costs in the EU (De Cara *et al.* 2005). The second approach uses supply side equilibrium models, where prices are endogenous. These models depict how a bigger region's economy or a particular sector of it would behave given the mitigation constraints, like the DICE and RICE models which encompass all major sectors of the global economy (Nordhaus and Boyer 1999), the ASMGHG model depicting the US agricultural and forestry sector (Schneider *et al.* 2007), and the CAPRI model of European agriculture (Dominguez *et al.* 2009). Finally, in the third group, engineering MACCs compile information on the costs and mitigation effectiveness of a set of MMs to calculate their average CE and then plot these MMs according to increasing CE to derive the MACC. Examples include the McKinsey MAC curves (Naucler and Enkvist 2009) and the sectoral UK MACCs commissioned by the Committee on Climate Change (Collier *et al.* 2013b). Typical features of the different MACCs are summarised in Table 1. Equilibrium

MACCs capture economy-wide interactions and is less prone to double counting of emissions or mitigation than the micro-economic and engineering approach, therefore well suited to answer international and cross-economy policy questions. On the other hand, the micro-economic and engineering approaches are better suited to explore the details of MMs (Kesicki and Strachan 2011) and advise regional and sector-specific policy development.

*Table 1 Typical characteristics of the three MACC approaches*

	Micro-economic MACCs	Equilibrium (or top-down ) MACCs	Engineering (or bottom-up) MACCs
Modelling approach	Econometric models	General equilibrium / partial equilibrium models	CE of individual MMs based on unitary costs and abatement
Spatial resolution	Sub-national regions	Sub-national regions	Country
Spatial coverage	Single region to groups of countries	Groups of countries	Country
Derivation of marginal abatement curve	Agents react to carbon tax by deploying a combination of MMs	Agents react to carbon tax by deploying a combination of MMs	Agents deploy MMs one after another, ordered by CE (interactions considered)
Shape of curve	Smooth	Smooth	Step-wise
Uptake rate of MMs	Endogenous	Endogenous	Exogenous
Negative CE	No negative costs	No negative costs	Costs can be negative (savings)
Prices	Exogenous	Endogenous	Exogenous
Representation of MMs	Detailed	Low level of details	Detailed
Economic interactions between sectors	Not considered	Included	Not considered

An illustrative example of an engineering MACC is presented in Figure 2. The MMs are represented as bars: the height of a bar shows the CE (i.e. the cost of the MM divided by the GHG abatement achieved), while the width shows how much abatement can be achieved. Consequently, the area of each bar corresponds to the total cost of the MM. Although the height of the bar represents the average CE of a single MM, at the same time it also reflects the marginal abatement cost at the level of total abatement through its position on the x axis, given that possible interactions between the MMs are included. In this example, the marginal benefit of mitigation is represented by the damage cost used by the UK Government in policy appraisal: £36 t CO<sub>2</sub>e<sup>-1</sup> (DECC 2009). The optimal abatement is 9 Mt CO<sub>2</sub>e, and it could be achieved by implementing the MMs on the left from the optimal abatement point. The MMs with negative CE (under the y axis) would provide financial savings if implemented, while those MMs with a CE between 0 and the carbon damage cost would cost

money but would provide larger marginal benefits than their marginal costs. The MMs to the right of the optimal abatement are estimated to be not economical to implement.

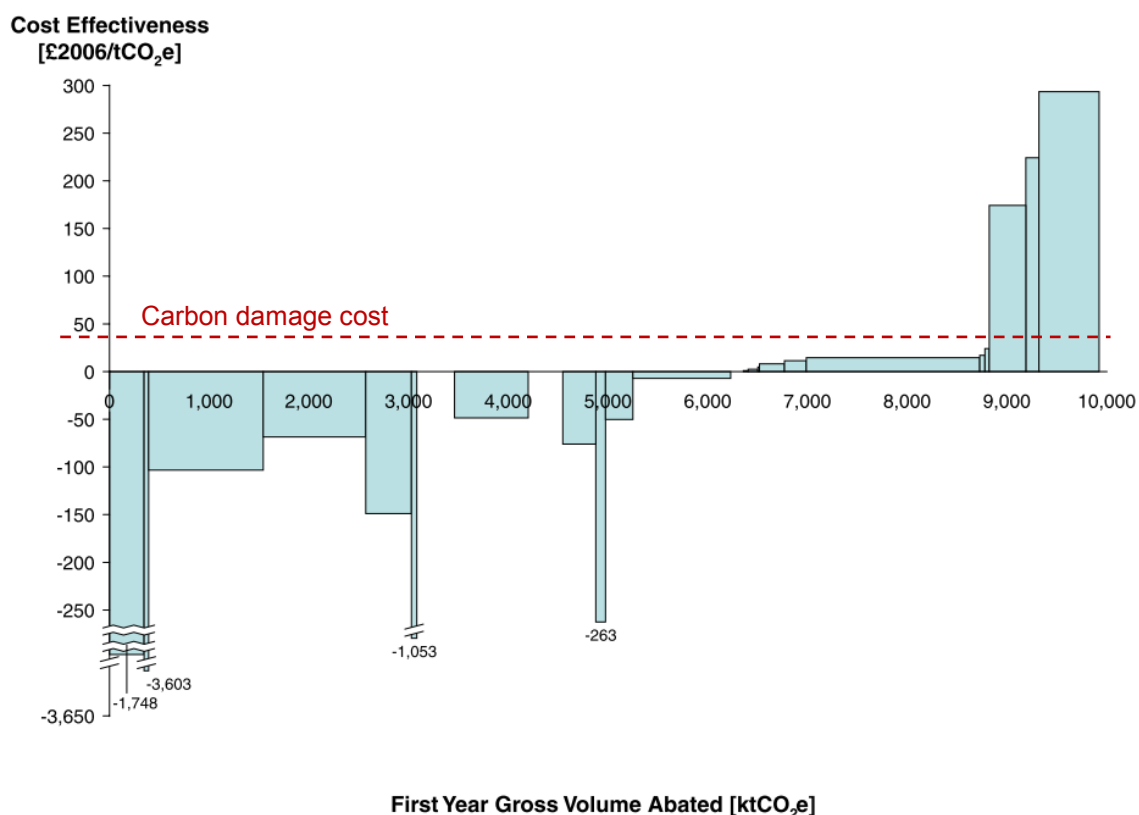


Figure 2. An illustrative MACC

*This curve shows the total UK agriculture, land use and land use change abatement, central feasible potential (CFP), 2022 (discount rate = 3.5%, measures with CE > 1,000 are not shown); for further information see (Moran et al. 2011b)*

### 1.3 Addressing the limitations of the MACCs

As any assessment tool, MACCs have their shortcomings. This section provides an inventory of the main limitations. Many of these issues have been discussed in the scientific literature (Kesicki and Ekins 2012, Kesicki and Strachan 2011, Vermont and De Cara 2010), and research has been carried out to address most of them. Nevertheless, many limitations are addressed only sporadically and the methodologies developed have not been taken up widely by subsequent studies. Table 2 provides a brief overview of the limitations and relevant – where possible agricultural – research targeting these limitations.

Table 2. Main limitations of the MACCs and examples for research targeting these issues

Main limitations <sup>1</sup>	Examples of studies targeting this limitation	Suggested further research	Category <sup>2</sup>
<b><u>Boundaries of the analysis are not fit for purpose or not clearly defined</u></b>	Some studies present MACCs with contrasting boundaries (Schulte <i>et al.</i> 2012).	MACCs and CE estimates should be distinguished and estimated at the farm level, domestic and global supply chain level wherever possible. See Chapter 6.	**
<u>Definitions of the MMs</u> are not specific enough at the farm level	Though some multi-sectoral and global agricultural MACCs assess very broad mitigation options, most sectoral MACCs are more specific.	More accurate MM definitions during research and intensive dialogue between researchers and stakeholders about the technological details would be beneficial.	*
<u>Discount rate</u> used is not fit for purpose	Some studies estimate the CE and the marginal abatement cost at different discount rates (Moran <i>et al.</i> 2008, Pape <i>et al.</i> 2008).	Social and private discount rates should be both used to create contrasting MACCs wherever possible.	**
<u>GHG effects</u> are not fully represented	N <sub>2</sub> O and CH <sub>4</sub> sources are considered in most studies, while CO <sub>2</sub> emissions and soil and biomass C sequestration are not always. A few studies include all sources mentioned (Golub <i>et al.</i> 2009, Schneider <i>et al.</i> 2007).	Even though the main agricultural GHGs are N <sub>2</sub> O and CH <sub>4</sub> , CO <sub>2</sub> emissions and C sequestration should both be mainstreamed in MACCs.	**
<b><u>Heterogeneity is not represented</u></b>	Most engineering MACCs are constructed for a region or a country aggregating all agents within a sector, but example exist looking at the heterogeneity of CE (Biggar <i>et al.</i> 2013)	Heterogeneity both in unitary abatement and costs are important factors in the potential uptake and total abatement estimates. MACCs for representative farm types, farm sizes or regions can be constructed to reveal the heterogeneity.	**
<u>Interactions</u> between the MMs and their effects on abatement and cost is not represented or not clearly defined	Most studies consider interactions to various extent, though they are not always clearly explained.	Clarity on the interaction methodology is needed. Furthermore, biophysical information on interactions could be used more widely in MACCs.	*

<sup>1</sup> Frameworks are developed in this dissertation to address the limitations in bold

<sup>2</sup> \*: most of the MACCs are adequate in this respect, but clearer reporting and communication of the relevant limitations to policy makers is needed;

\*\*: some MACCs overcome this limitation, but wider uptake of these approaches (when appropriate to the policy question) is needed;

\*\*\*: no agricultural MACC has addressed this problem appropriately, according to the knowledge of the author

Table 2. cont.

Main limitations <sup>1</sup>	Examples of studies targeting this limitation	Suggested further research	Category <sup>2</sup>
<u>Marginal benefits</u> are misrepresented	Studies report the whole MACC curve and thus abatement potential at various marginal benefit values (i.e. CE threshold) can be obtained.	MACC studies should provide the spatially and temporally relevant marginal benefit value.	*
<u>Market effects</u> are not represented in engineering and micro-economic MACCs	Hybrid approaches (a combination of equilibrium and engineering models) exist in other sectors (Andreas Schafer and Henry 2006).	The high differences in abatement rate between equilibrium models on one side and micro-economic and engineering models on the other side (Vermont and De Cara 2010) suggest that a hybrid approach in agriculture would be informative.	***
<u>Non-monetary barriers</u> are not represented	No examples were found.	No study seems to have addressed the non-monetary barriers explicitly in agricultural MACCs. Agent based modelling and multi-criteria analysis might be useful approaches to complement MACCs in this respect.	***
<b><u>Transaction costs are not represented, CE of policy instruments (PIs) are not investigated</u></b>	Transaction cost studies exist regarding agro-environmental policies (Ducos <i>et al.</i> 2009, Krutilla 2011, Mettepenningen <i>et al.</i> 2009).	No agricultural MACC seems to have explicitly taken into account transaction costs or explored the CE of whole PIs; transaction costs depending on the PI suggested should be part of the total costs. See Chapter 4.	***
<b><u>Wider effects are not represented</u></b>	MACCs showing some co-effects in physical quantities already exist (Anthony <i>et al.</i> 2008, Brink <i>et al.</i> 2001, Brink <i>et al.</i> 2005, Wagner <i>et al.</i> 2012), and non-agricultural studies exist on monetised co-effects (Gielen and Changhong 2001).	No study seems to have included the monetised co-effects into agricultural MACCs; the wider effects should be part of the CE assessment. See Chapter 3.	**
<b><u>Uncertainty is not represented</u></b>	Examples of MACC with uncertainty analysis exist in other sectors and whole economy assessments (2006).	Agricultural MACCs should also include uncertainty analysis. See Chapter 5.	***

<sup>1</sup> Frameworks are developed in this dissertation to address the limitations in bold

<sup>2</sup> \*: most of the MACCs are adequate in this respect, but clearer reporting and communication of the relevant limitations to policy makers is needed;

\*\*: some MACCs overcome this limitation, but wider uptake of these approaches (when appropriate to the policy question) is needed;

\*\*\*: no agricultural MACC has addressed this problem appropriately, according to the knowledge of the author



As Table 2 shows, some of these issues have been tackled by a number of authors, while others were hardly, or not at all addressed in the case of agriculture. Three potential problems have been overcome in most agricultural MACCs; these are inaccurate MM definitions, accounting for interactions and using appropriate marginal benefits. However, reporting and communicating the methodology and its limitations to stakeholders still need to improve in these areas. Five potential limitations (boundaries of the analysis, choice of discount rate, accounting for all main GHG effects, heterogeneity and considering wider effects) have been addressed at least in one study about agricultural MACCs; these methodologies are potentially transferable. A wider future use of these approaches is suggested. Finally, four more limitations (inclusion of market effects, non-monetary barriers, transaction costs / policy CE and the accounting for uncertainty) have not been addressed in agricultural engineering MACCs to the knowledge of the author. Here a greater research effort is required in the future.

Methodological improvements have been made in this dissertation to two out of the four limitations that have not yet been addressed by other authors (transaction costs / policy CE and uncertainty). The other two have not been targeted due to lack of data availability (non-monetary barriers) and limitations in time available for modelling development (market effects). Three more limitations have been selected for further improvements amongst those where some authors have already made attempts to widen the MACC methods: the inclusion of wider effects, considering heterogeneity and the expansion of the boundaries of the mitigation. The following sections provide a background on these limitations.

### 1.3.1 Wider effects

MACCs are designed to look at the CE of reducing one specific type of externality. As opposed to cost benefit analysis (CBA) they have the advantage of looking at the pollution in physical units instead of converting these units into financial units. This prevents introducing an additional uncertainty related to monetising the effects of the pollution. On the other hand, this also constrains the analysis to that single pollutant, without offering an easy way to compare the GHG mitigation efforts with actions to abate other pollutants. Furthermore, the use of physical units does not take into account the effects of GHG mitigation efforts on other pollutants. However, the wider effects can significantly change the results of CE or CBA assessments (Glenk and Colombo 2011, Nemet *et al.* 2010).

Not including the wider effects can become a limitation for two reasons. First, a MACC cannot be used to answer questions about the most efficient allocation of funds among different environmental goals. However, the MACC approach is still well suited for high-priority issues, like climate change, or where previously agreed pollution thresholds have to be achieved (e.g. regional water pollution), or when the funding sources for the particular pollutant have already been agreed upon.

Second, if a GHG mitigation activity affects other environmental goals either in a positive or in a negative way, for example reducing GHG emissions but at the same time decreasing diffuse water pollution or increasing food scarcity, the co-effects would make the GHG mitigation activity more or less desirable than a pure GHG CE metric can tell us. Many potential mitigation activities in agriculture have significant co-effects on air pollution ( $\text{NH}_3$ ), diffuse water pollution (nitrate ( $\text{NO}_3^-$ ) leaching), biodiversity and food safety. Assessing these effects together is of high importance. For this purpose a single pollutant MACC can be complemented with a qualitative or quantitative assessment of the co-effects, thus providing policy guidance on the overall effects of the MMs.

The research on modelling the effects of multiple pollutants has been developing rapidly in the past two decades. There are two divergent technical solutions for the integration. The pollutants can be represented in one single model, as in the GAINS model (Amann *et al.* 2011), where five air pollutants and six greenhouse gases are considered. Alternatively, the effects on different pollutants can be modelled independently, like in Anthony *et al.* (2008), who used six different process models to obtain information on six pollutants. The single model approach might require more investment in model development but can provide better consistency and easier future application, while the benefits of the other approach is that it can include more detailed and robust results on the individual pollutants.

There are two main approaches for the optimisation as well. One method is to optimise for one pollutant while presenting the effects on the other pollutants, see an example by Schneider *et al.* (2007). The other was is looking for optimal solutions integrating the effects of all pollutants in parallel (Anthony *et al.* 2008). This integration can be achieved in three ways. First, if a common pollution unit can be derived for the pollutants in question, a simple MACC can be constructed. This is the case for all GHG MACCs which look at more than one GHGs: the common metric is  $\text{CO}_2$ -equivalents; non- $\text{CO}_2$  gases usually being converted by using global warming potential (GWP) values. Alternative metrics are also in use, such as radiative forcing (van Vuuren *et al.* 2006). The choice of metric makes a difference in the

importance of the different GHGs over short and long time horizons (Reilly *et al.* 1999). Prioritising MMs within agriculture and between agriculture and other sectors has to take into account this issue, as the majority of agricultural GHG emissions both globally and in Europe are in the form of non-CO<sub>2</sub> gases CH<sub>4</sub> and N<sub>2</sub>O.

Second, when no physical unit can be easily constructed for the integration of different pollutants, a composite indicator can be constructed (OECD 2008). To do so, the effects of the various pollutants have to be normalised in order to allow comparison (e.g. by comparing each to a respective target, like a damage threshold, or, if such a target value is not available, using standardisation or min-max techniques). Weighting and aggregation rules also have to be set. Preferential weighting of the pollutants (and other indicators, including social targets and costs) can be developed in a multi-criteria analysis (MCA) approach, where the importance of each indicator in the evaluation is set by the decision makers or the analysts (Linkov *et al.* 2006). The approach allows for including effects which only have semi-quantitative information and is well-suited to reflect stakeholders' preferences. MCA has been used in the assessment of GHG mitigation PIs (Konidari and Mavrakis 2007) and adaptation strategies (de Bruin *et al.* 2009).

Finally, the effects on multiple pollutants can be integrated via converting the physical units to monetary values (Winiwarter and Klimont 2011). This is possible if the damage cost estimates of the pollutant are available. The monetary value of the damage avoided can be added to the financial costs of the MM and then evaluated against the primary pollution thus conducting a CE analysis extended to co-effects. On the other hand, if all the environmental effects are converted into monetary terms, a cost benefit analysis becomes possible (Pretty *et al.* 2000). While the results of such approaches can be presented in visual ways which are easy to understand, the choice of damage values might have a significant impact on the results. This can limit the usefulness of the method particularly when the damage values are very uncertain, have a high spatial or temporal variability or if a strong threshold effect exists.

Chapters 2 and 3 are exploring environmental co-effects of pollution reduction efforts. The GHG co-effects of NH<sub>3</sub> emission reduction are discussed in Chapter 2, while Chapter 3 presents a MACC framework possible of integrating the co-effects into MACCs.

### 1.3.2 Transaction costs and the cost-effectiveness of policy instruments

Agriculture, consisting of a very heterogeneous group of agents and burdened with the difficulties of spatially and temporally highly variable GHG emissions, is a sector where market-based instruments are usually very costly to set up. This, combined with other barriers in international relationships, creates a situation where promoting mitigation via voluntary or targeted obligatory regulations are the favoured PIs over market-based instruments (Beddington *et al.* 2012, Kasterine and Vanzetti 2010). The development of such PIs, particularly the prioritisation of MMs for compulsory and voluntary regulations, requires detailed information on the MMs. MACCs derived from equilibrium models provide information for the evaluation of general policy scenarios (e.g. a carbon tax or a subsidy), but are less suitable to the comparative analysis of MMs, which, in turn, could feed into regional policy development. Engineering MACCs are capable of informing this type of policy development well, though two considerations have to be addressed.

The effectiveness of policies in terms of generating additional abatement is an important factor in policy CE. PIs operating on the basis of non-subsidised voluntary uptake are likely to achieve lower uptake than PIs providing financial subsidies and thus transferring part of the private costs to public costs. Moreover, compulsory regulations might lead to even higher uptake, though often at increased transaction costs of monitoring and enforcement. Engineering MACCs work on a basis of an assumed uptake rate, which has a major influence on the abatement achievable. In some MACCs 100% uptake is assumed, i.e. the results present the maximum technically available abatement (Moran *et al.* 2008), while other MACCs divide the 100% uptake evenly between the measures (DeAngelo *et al.* 2006), or they assign uptake values to individual MMs (Pellerin *et al.* 2013). However, rarely do they link their assumptions directly to PIs to be used in the future. Such assumptions on policy compliance can be derived from *ex post* assessments of similar environmental policies (Mettepenningen *et al.* 2009) or can be estimated via econometric models (Ducos *et al.* 2009). Additionally, the level by which public agents take over cost elements from the private sector has a profound effect on uptake, and for this reason it is good practice to make a distinction between private and public costs in the MACCs.

Second, transaction costs, which go beyond the technical implementation of the MMs, can be of high importance. These are the costs related to policy making from the planning phase

through enforcement, and can range between 21-50% of total costs (Coggan *et al.* 2010). They include *ex ante* costs of establishing environmental entitlements (e.g. information collection, legislation development) and *ex post* costs of implementation (e.g. administration, contracting, monitoring, enforcement) (Krutilla 2011, McCann *et al.* 2005). Although most of these costs are usually public costs, private agents also incur part of them mainly in the form of time required for information gathering and record keeping. Estimating these costs is often difficult, but including them in the calculations can improve the abatement and cost projections.

Overall, an engineering MACC can be used to inform *ex ante* assessment of the CE of environmental policies by first producing a maximum technical potential abatement MACC and then building in information on policy packages (i.e. clearly defined PIs targeting a detailed set of MMs, with an estimated level of uptake and compliance, complemented with transaction costs estimates).

Policy tools targeting packages of MMs are discussed in Chapter 4, introducing a framework to calculate the CE of PIs through a MACC analysis. The case study focuses on Scotland.

### 1.3.3 Uncertainty

Robust policies, which are able to achieve their objectives across a range of possible futures, have to be developed by taking into account uncertainties (Lempert and Schlesinger 2000). Uncertainty analysis is becoming part of economic assessments looking at GHG mitigation, particularly in global, multi-sector models and in the energy sector (Peterson 2006). However, to date, research on the economics of GHG mitigation in agriculture has rarely included uncertainty analysis, even though this would be of high importance to inform regional, MM specific policy design. The heterogeneity of the sector and the spatial and temporal variability of emissions pose particular challenges for uncertainty assessments.

Uncertainties associated with uptake levels, mitigation potential and costs of future GHG MMs all contribute to the uncertainty of the MACC. They are a result of both the stochastic nature of and our limited knowledge about the underlying biogeochemical, economic and societal processes, human behaviour and politics. On one side, biogeochemical processes have a significant influence on land use activities, farm management decisions and associated emissions. Their uncertainties filter through to the uncertainties in the effectiveness of MMs. One example is the N<sub>2</sub>O emissions arising from N fertiliser

application. These emissions vary significantly with the weather conditions under static management, and at the same time weather conditions also define management decisions about fertilisation, adding another layer of stochasticity to the GHG emission and mitigation. On the other side, the economic and policy environment are also crucial in land use decisions, and their uncertainty contributes to the MACC uncertainty as well. For example price fluctuations, the uncertainty in future changes in policy regulations (e.g. subsidies for renewable energy or Common Agricultural Policy (CAP) payments) and farmers' reaction to these changes are all important factors in uncertainty.

Some of these uncertainties can be quantified (expressed as probabilities) and hence included in quantitative models, although information on them often does not yet exist in the scientific literature. Other uncertainties cannot be quantified statistically; this is particularly the case for complex models predicting the future (Hallegatte *et al.* 2012) and for value uncertainty, like the choice of discount rate.

The MACC methodology is able to accommodate information on quantifiable uncertainty of the optimal abatement and the CE of the MMs and can convey it in a relatively simple language. Uncertainties which cannot be quantified have to be presented alongside the results as well so that policy makers and other users of the MACCs would be fully aware of the applicability and robustness of the results. Not overlooking the difficulties of communicating uncertainty between scientists and policy makers, a mutual engagement from both sides is required to widen the use of this type of information in policy making (Smith and Stern 2011).

Chapter 5 discusses the uncertainties in the agricultural MACCs, reports on a MACC framework suitable for uncertainty analysis of the MACC and presents the results of an application on the Scottish agricultural MACC.

### 1.3.4 Boundaries

MACCs relate to a sector or the whole economy of a region, country or group of countries, to a particular time period, and they also have boundaries in terms of what cost elements and emission sources are included. According to these features they can be useful in different policy contexts.

MACCs depicting national mitigation effort in a group of countries can inform international agreements and regulations, like policies in the European Union (Blok *et al.* 2001). Similarly, a series of sectoral MACCs related to an economy can be used for designing sector-specific targets, e.g. the UK Carbon Budgets (Anon. 2014). However, to avoid double counting of mitigation, the boundaries among the set of MACCs have to be clearly defined and stay within their respective regional and sectoral limits. In such a methodology MMs are evaluated within the relevant spatial/sectoral boundaries without considering effects beyond. For example, reducing the N content in livestock diets might reduce the emissions related to soya production. Similarly, if less synthetic N fertilisers is applied, the emissions from N fertiliser production are also mitigated. These effects occur out of the farm gate. Taking into account the full GHG effects of MMs are possible with a life cycle analysis (LCA) approach. As Schulte *et al.* (2012) present, the resulting abatement potential and CE can be different from the conventional MACC. While the national and sectoral approach is important for allocating mitigation effort between countries and sectors, decision makers should not solely rely on them. The assessment should be complemented with LCA-based results to avoid potential emission leakage, where unintended additional emissions happen in other countries or in other parts of the supply chain.

A related issue is the methodology used to calculate the baseline GHG emissions and the potential mitigation, particularly the difference between IPCC methodology and other approaches. The Annex I countries calculate their national GHG inventories by a combination of methodologies called Tier 1, Tier 2 or Tier 3 methods (IPCC 2006), what considers mitigation where there is a robust evidence to justify the effect. As national inventories provide the starting point for international agreements, policy makers usually aim to achieve mitigation which can be reflected in these inventories. On the other hand, MACCs explore further MMs which currently might lack robust, quantified evidence on the overall mitigation effect. Furthermore, the abatement potential of an MM is different depending on which IPCC methodology is used, and, as discussed also above, differs further with other methodologies (Lengers and Britz 2012). Therefore, policy makers may find it useful to know how much of the abatement can be represented in the national inventory, what methodological developments are needed to reflect all robust mitigation in the national inventory, and what the additional emission consequences are of those MMs which are not represented on a particular MACC or in the national inventory.

A similar dilemma exists for the choice of production and consumption based emission accounting. Consumption based emissions differ from production based emissions in that they include the embedded emissions of imported goods and services and exclude the emissions related to the production of exported goods and services. Production related emissions are higher than consumption related emissions in China and Russia, while most of the Kyoto Protocol Annex I countries show the opposite pattern (Lenzen *et al.* 2013). The gap between consumption and production based emissions have been widening in the past years in the UK (Collier *et al.* 2013a). Most agricultural MACCs have been using a production based accounting. Even those, which consider one or more MMs targeting food consumption, look at production based emissions (Westhoek *et al.* 2014), and do not account for imports and exports of agricultural products. The production based approach is suitable for most of the policy options as these policies mostly target the farmers rather than the consumers, but for the assessment of PIs targeting consumption further developments are needed in the MACCs.

A final boundary concern is the temporal relevance of the MACCs. Annual MACCs (or a series of annual MACCs over a time period) can be used as snapshots of abatement potential and costs when planning for reduction targets and milestones. However, cumulative MACCs (integrating abatement potential over a longer period of time) can be also useful if MMs differ significantly in how their costs, mitigation efficiency or uptake changes over time. While shorter-term MACCs, limited to individual sectors and regions, can feed into rapid policy design, longer term and economy-wide MACCs have also be considered to avoid lock-in situations. Lock-in situation can occur if a pathway, which is favourable in the short-term, is followed and makes it costly to change to another set of actions, which are more favourable in the long-term (Vogt-Schilb and Hallegatte 2011). Even though many of the MMs suggested in agriculture are easily reversible (e.g. the administration of animal feed additives), longer term investments like establishing or improving soil drainage, irrigation, animal housing or anaerobic digestion might create pathway-dependency. These can become obstacles when system transitions (e.g. changing the location of cropping and grazing areas) become preferable (especially due to changing climatic conditions).

A modelling framework to assess the farm-level CE, including emission changes achieved outwith of the farm is introduced in Chapter 6, describing a way how emission displacement within the supply chain can be avoided. The case study is the wider application of sexed semen in dairy herd management in Scotland.



### 1.3.5 Heterogeneity

The heterogeneity in the farming system (regarding farm types, like dairy versus cattle farms, farming practices, like grass-based versus indoor cattle systems or solid versus liquid based manure storage systems, climatic and soil conditions and farmer behaviour) is reflected in the difference in the pollutant profiles of the various farms (Dalgaard *et al.* 2011). Beyond the heterogeneity in emissions, the sector also exhibits heterogeneity in resource constraints (e.g. access to labour and capital) and in the financial structure of the farm. Altogether, these result in the heterogeneity of abatement potential and abatement costs. As engineering MACCs use estimated average values to describe the sector, the range of potential differences in abatement and CE of the MMs is most often overlooked; one of the few exemptions is the CE analysis of mitigation options in the US agriculture (Biggar *et al.* 2013). Micro-economic and equilibrium models represent a range of farms, farm types or agricultural regions and therefore are able to present the heterogeneity of the abatement costs (De Cara and Jayet 2000), though not for individual MMs. Without information on the level of heterogeneity and the differences between farm types, farm sizes, etc. the effectiveness of PIs targeting specific MMs might be lower than expected.

A modelling framework capable of addressing heterogeneity of the CE of MMs is introduced in Chapter 6, presenting an analysis of two dairy farm types in Scotland.

### 1.3.6 Other limitations

This section briefly discusses those limitations which were introduced in Table 2 but are not specifically addressed in this dissertation.

#### 1.3.6.1 DEFINITIONS OF THE MITIGATION MEASURES

Engineering MACCs are based on the assessment of technological or management options, which are assumed to be replacing current technology or management and thus provide MM. Due to the differences in farming practices, the categorization of the MMs are not without difficulties. Moreover, scientific reports and knowledge exchange documents all differ in terms of the broadness and accuracy of their MM definitions. For example ‘nutrient management’ can be considered as one broad MM, but often it is separated into five, ten, or more MMs, like ‘Reduce the rate of mineral fertiliser by more effectively adjusting yield targets’, ‘Improved timing of mineral fertiliser N application’ or ‘Use nitrification

inhibitors'. These MMs are sometimes further divided, like the 'Use nitrification inhibitors' can be disaggregated according to the type of fertiliser it is used on (mineral or organic). Though the context specific development of MMs is essential and a universal MM list do not exist (Smith 2011), the diversity in the definitions makes the comparison between studies difficult.

A further complication is that very often the choice between practices is more of a continuous than of a discrete nature. A typical example is the timing of nitrogen fertilisation. Though a discrete choice exists between applying the whole amount of fertiliser at once as opposed to splitting it into two (or more) applications, the exact timing of the application relative to the growth stage of the crop is almost a continuous choice. Similarly, changing the proportion of ingredients in the livestock diet is a continuous choice. For instance, 'Increasing concentrates in the diet' is widely featured as a MM, but studies very rarely provide detailed advice on what should be the composition of these concentrates and what proportion should they make up in the diet. The choices and circumstances in the previous examples are all important driving factors for GHG emissions, but cannot be easily defined and described. These – practically necessary – simplifications increase the uncertainty in the abatement potential estimates and at the same time enhance the risk of miscommunication between researchers and stakeholders, particularly farmers.

#### 1.3.6.2 DISCOUNT RATE

The choice of discount rate is a much debated and very important question in environmental CBA and cost-effectiveness analysis (CEA). The discount rate reflects the time preference of the individuals and society: the higher the discount rate, the more emphasis is put on costs and income happening earlier. For private investments normally the return on investments is used as discount rate, representing how much the money could grow if invested in the market. Contrastingly, long-term public investments are discounted with a lower discount rate to represent inter-generational equity better. This social discount rate can be constructed as a declining discount rate as opposed to a constant low discount rate (Arrow *et al.* 2013). Individuals might also have discount rate preferences depending on the time frame considered (Grijalva *et al.* 2014). The existence of these two contrasting discount rates, private and social, poses a problem to MACC analysis: for example, shall afforestation costs and benefits be discounted with a private or a social discount rate? Which discount rate shall be used when assessing the installation of anaerobic digesters? The answers would partially depend on whether the investment is expected to be financed by private individuals or from

public money. As MACCs often do not refer to a very particular policy environment where the share of public *versus* private funding is determined for each MM, the choice of discount rate remains unresolved. However, alternative MACC evaluations can be presented based on the different discount rates and this can inform policy makers about how the CE of the MMs are different from the perspective of the farmers and of the public budget.

#### 1.3.6.3 ACCOUNTING FOR ALL GHG EFFECTS

As discussed in Section 1.1, agricultural GHG emissions consist of N<sub>2</sub>O, CH<sub>4</sub>, and only to a smaller extent of CO<sub>2</sub>. On the other hand the relative contribution of agriculture to the total N<sub>2</sub>O and CH<sub>4</sub> emissions is high. Consequently, the main focus of MACC studies has been on N<sub>2</sub>O and CH<sub>4</sub> mitigation, with little or no attention to CO<sub>2</sub> emissions and C sequestration. This might be unintentionally encouraged by the IPCC GHG inventory methodology, where some important CO<sub>2</sub> sources and CO<sub>2</sub> sinks (namely emissions and sequestration from agricultural land use change and emissions from fuel and energy use) are reported outside of the ‘Agriculture’ category.

While by now most agricultural MACCs consider CO<sub>2</sub> as well as N<sub>2</sub>O and CH<sub>4</sub>, changes in biomass and soil C content are often neglected, even though globally, the majority of the economically efficient abatement potential arises from the increase in soil C stocks (Smith 2011). This suggests that the soil and biomass C stock changes are of importance for MACC studies, and neglecting them might result in underestimating the abatement potential.

#### 1.3.6.4 INTERACTIONS

MMs often involve making management or infrastructural changes on the same production factors or processes. To construct a MACC, this necessitates the MMs to be considered as processes interacting with each other, both in terms of mitigation effect and costs. Assessing the MMs independent of each other leads to a so-called stand alone CE analysis of the MMs, which is informative if it is likely that agents will only implement one or very few MMs. However, stand alone assessment is not suitable for deriving a cumulative abatement potential due to potential double counting, or not accounting for mitigation and costs. MACCs based on equilibrium models and micro-economic models inherently capture part of these interactions, based on how the GHG emissions are represented in the models. Those interactions which relate to resource use belong to this group, for example the N<sub>2</sub>O mitigation potential which can be achieved by optimised N fertilisation of grasslands is reduced if the share of legumes in the swards has been increased – this is usually captured in

the models if the fertilisation rate on grass-legume swards is lower than on pure grass swards. Other interactions can only be included in these models by additional modifications in the model, for example if the N flow is not built in the model, then the reduced N<sub>2</sub>O abatement achievable with slurry cooling having already decreased the N content of the diet is not automatically modelled. On the other hand, engineering MACCs do not include any interactions by default, but they have to be built in them via interaction factors which alter the abatement potential or costs of a MM based on which other MMs have already been implemented (i.e. all of the MMs which are to the left on the MACC from the MM in question).

#### 1.3.6.5 MARGINAL BENEFITS OF MITIGATION

To attain the economically optimal abatement the marginal costs of GHG mitigation have to be compared with the marginal benefits. GHG mitigation benefits arise from the avoided impacts on society resulting from changes in e.g. global temperature, extreme weather events, sea level rise. The marginal benefits are a function of the emission level, i.e. the amount of emission abated (Tol 2005), the location of the benefits (Anthoff *et al.* 2009), and the timing of the mitigation (Frankhauser and Tol 1996). Though these considerations are important in long-term and international decisions, they are rarely considered in MACCs, where usually the marginal benefit is approximated with a constant. This simplification is partly appropriate for MACCs carried out at a smaller scale, e.g. at national level, assuming that the mitigation level does not have a significant effect on the global emissions – this assumption corresponds with the fact that policy decisions about mitigation efforts are ultimately determined at a national level (Anthoff and Tol 2010). However, marginal benefits can change considerably over time, for example Price *et al.*'s (2007) estimation for the shadow price of carbon changes from £18.6 t CO<sub>2</sub>e<sup>-1</sup> in 2000 to £59.6 t CO<sub>2</sub>e<sup>-1</sup> in 2050. Thus the economically efficient abatement potential can change considerably over time as well. A representation of this temporal change is possible by constructing a set of annual MACCs as snapshots covering a time period, each of them using the relevant carbon price.

#### 1.3.6.6 MARKET EFFECTS

Agri-environmental policies targeting GHG emissions are likely to impose various changes in the costs of crop and livestock production, which, in turn, might lead to changes in the supply of agricultural products, especially if the policy impacts on a substantial number of farms. A shift in the supply will eventually lead to changes in the agricultural commodity

prices, with a feedback on the whole sector and beyond. By definition, micro-economic and engineering MACCs are not capturing these feedback loops, and therefore their estimates are only valid within the assumption that prices will not be significantly affected by the mitigation policies. On the other hand, in equilibrium models these prices are endogenously defined, and their results account for the market effects.

#### 1.3.6.7 NON-MONETARY BARRIERS

MACCs capture the technological costs, for example investment in new machinery and savings in resource use. As mentioned above, other cost elements, like transaction costs and policy costs are usually not considered, and, as they are based on some form of profit maximisation assuming rational agents, neither are behavioural barriers included. In reality, farmers have a mixture of objectives which include profit maximisation, but also risk aversion, environmental attitudes and social context as important factors in decision making (Pannell *et al.* 2006). These factors, along with lack of information, regulatory and market constraints, can create barriers for uptake of MMs beyond the cost aspects (Feliciano *et al.* 2014). This phenomenon is most visible in the presence of ‘win-win’ MMs on the engineering MACCs. Win-win MMs are estimated to have negative costs and CE, i.e. they are estimated to generate savings. Assuming rational agents the win-win measures can only be understood as a consequence of underestimated costs, but the existence of non-monetary barriers can explain their appearance on the MACCs. When policy makers consider the low hanging fruits, or, indeed, any other MM, information on these barriers is needed to complement the CE results in order to design efficient policies.

### 1.4 Structure of the dissertation

The following chapters present the frameworks developed to address the above mentioned four limitations, accompanied with a case study application of each approach and a discussion of the particular limitation and the results of the case study.

The co-effects on GHG emissions of reducing NH<sub>3</sub> emissions from agriculture are discussed in details in Chapter 2, while a possible way to integrate other pollutants into GHG MACCs is presented in Chapter 3; both chapters addressing the problems of integrated assessment to explore the environmental co-effects of pollution reduction efforts.

The omission of wider effects of GHG mitigation has been highlighted as a drawback of GHG MACC analysis. This issue is serious in agriculture, where  $\text{N}_2\text{O}$  emissions have complex interactions with other forms of reactive nitrogen ( $\text{N}_\text{r}$ ) (e.g.  $\text{NH}_3$  and  $\text{NO}_3^-$ ), and management decisions affect the balance between the three main agricultural GHGs. A review of in Chapter 2  $\text{NH}_3$  MMs highlights the synergies and trade-offs between  $\text{NH}_3$  and GHG emissions. The most promising win-win measures are improving production efficiency, in particular improving N-use efficiency, along with low-emission livestock housing design and management, slurry acidification and urease inhibitors. On the other hand, separating slurry or increasing the aeration of solid manure, might result in increased emissions in GHGs.

A multiple-pollutant cost-effectiveness analysis in Chapter 3 shows how the quantitative inclusion of external effects ( $\text{NO}_3^-$ ,  $\text{NH}_3$ , phosphorous (P) and sediment) can alter the cost-effectiveness of GHG MMs and how the economically optimal abatement potential changes with alternative damage cost estimates. Higher damage cost values improve both the cumulative GHG abatement potential and the associated gains in the other four pollutants. Using lower damage costs the total cost of pollution is dominated by GHG emissions, while very high damage costs (20,577, 52,055, 45,144 and 108, £  $\text{t}^{-1}$ , for  $\text{NO}_3^-$ -N,  $\text{NH}_3$ -N, P and sediment, respectively) would justify the implementation of almost all MMs that have positive co-effects.

Chapter 4 elaborates on the CE of PIs, making the MACC analysis more relevant for policy design and offering a tool to integrate policy costs into the analysis. MACCs usually account only for the technical costs of the mitigation measures and omit the policy costs. Moreover, the assumed uptake rates are not explicitly linked to policy instruments. A policy cost-effectiveness assessment in Chapter 4 presents the challenges of translating ambitious technical mitigation potential from MACC analysis into cost-effective policy potential, analysing the policy packages suggested by the Scottish Government. These policies planned can deliver 383 kt carbon dioxide equivalent GHG saving in agriculture in 2022 – a 2.7% reduction from the 1990 baseline Scottish agricultural emissions. In contrast, the agricultural MACC for 2022 estimated the theoretical maximum potential to be of 2,584 kt  $\text{CO}_2\text{e}$ , and the cost-effective mitigation potential to be of 636 kt  $\text{CO}_2\text{e}$ .

Chapter 5 reports on the uncertainties in the optimal abatement and the ranking of the MMs, supporting robust policy design. Information on the uncertainty of quantitative results feeding into decision making is essential for designing robust policies. But this information

is often not available in relation to agricultural GHG MACCs. Chapter 6 presents an analysis of the uncertainty related to the Scottish agricultural MACC, offering a qualitative assessment identifying the different sources and types of uncertainty and a quantitative assessment estimating the statistical uncertainty of the MACC results. The results show that the uncertainty in the economically optimal abatement on Scottish agricultural land is high, however, the ranking of the measures is relatively robust, especially in terms of which measures have cost-effectiveness below the carbon price threshold.

A modelling framework to assess the farm-level CE, including emission changes achieved outwith of the farm is introduced in Chapter 6, describing a way how emission displacement within the supply chain can be avoided. GHG MMs on farm often have effects on emissions which arise elsewhere in the supply chain. To account for these effects an LCA approach is adopted in the GHG calculations in Chapter 6. Furthermore, a farm-based assessment allows for a distinction to be made between different farm types in terms of their potential for cost-effective abatement and most optimal MMs. The results of the case study on using sexed semen on dairy farms show that this measure might be a cost-effective way to reduce emissions from cattle production, noting that the GHG savings do not occur directly on the dairy farm, but elsewhere in the supply chain: in the beef production system.

The final chapter provides a conclusion focusing on policy use and proposes guidelines for policy makers on what kind of MACC would be the most informative depending on the policy context.





## 2 Paper I. Co-benefits and trade-offs between greenhouse gas and air pollutant emissions for measures reducing ammonia emissions and implications for costing

Vera Eory, Dominic Moran, Cairistiona F. E. Topp

Research Division, SRUC, West Mains Road, Edinburgh EH9 3JG, UK

### 2.1 Abstract

Both ammonia and greenhouse gases have been in the environmental research and policy spotlight in the past decades. Scientific evidence from the natural sciences and from economics have informed policy development and lead to different forms of regulations and policies both on ammonia and greenhouse gas emissions from agriculture, a sector which is an important source these pollutants. Moreover, the biophysical and management processes in agriculture create a situation whereby the emission of these gases are linked, commonly resulting in synergies and trade-offs in mitigation practices. An understanding of these synergies and trade-offs is key in designing efficient integrated policies. This chapter contributes to that effort by providing an overview of the greenhouse gas co-effects of some of the key ammonia mitigation measures.

Evidence suggests that some win-win solutions are available where both ammonia and greenhouse gas emissions can be reduced on farms; these include improving nitrogen use efficiency in livestock and crop production, low-emission livestock housing design, slurry acidification and urease inhibitors. Conversely, pollution swapping (trade-off between ammonia and greenhouse gas reduction) is likely to occur with ammonia mitigation in other cases, for example if the amount of starch and sugar in animal feeds is increased, if changes to housing and manure management systems are made, if slurry is separated to a solid and a liquid fraction or if solid manure is aerated during storage. The effects of some measures e.g. low-trajectory manure spreading, covering slurry stores and manure heaps, and anaerobic digestion of animal waste are currently uncertain and require further investigation.

## 2.2 Introduction

NH<sub>3</sub> pollution is one of many environmental burdens arising from human activities. Both globally and in Europe the main source of NH<sub>3</sub> emission is agriculture, particularly animal husbandry (European Environment Agency 2013, van Vuuren *et al.* 2011); cattle and swine populations contributed by 54% to NH<sub>3</sub> emissions in the EU-27 in 2011, while another 20% of emissions originated from synthetic N fertiliser use. In the same year, agriculture's share of EU-27 GHG emissions was 10%, mostly as N<sub>2</sub>O and CH<sub>4</sub> not including the CO<sub>2</sub> emissions and carbon sequestration effects of land use and land use change (European Environment Agency 2014). The agricultural emissions of CH<sub>4</sub>, N<sub>2</sub>O and NH<sub>3</sub> are interrelated: they have common sources and their emission rates depend on common factors, such as farm management, weather conditions and soil type.

N is an important element in agricultural production, and was the limiting factor in crop production before inorganic fertilisers became widespread (Smil 1999). N<sub>2</sub>O and NH<sub>3</sub> are parts of the N cascade, whereby the captured atmospheric di-nitrogen (N<sub>2</sub>) is transformed into various forms of reactive nitrogen (Galloway *et al.* 2003). Moreover, they can be transformed into each other in biochemical processes. The agricultural activities responsible for N<sub>2</sub>O, NH<sub>3</sub> and CH<sub>4</sub> emissions overlap; animal husbandry emitting NH<sub>3</sub>, N<sub>2</sub>O and CH<sub>4</sub> and crop production being responsible mainly for NH<sub>3</sub> and N<sub>2</sub>O emissions (Figure 3). This complex relationship between biophysical and management processes make synergies and trade-offs inherent in the system.

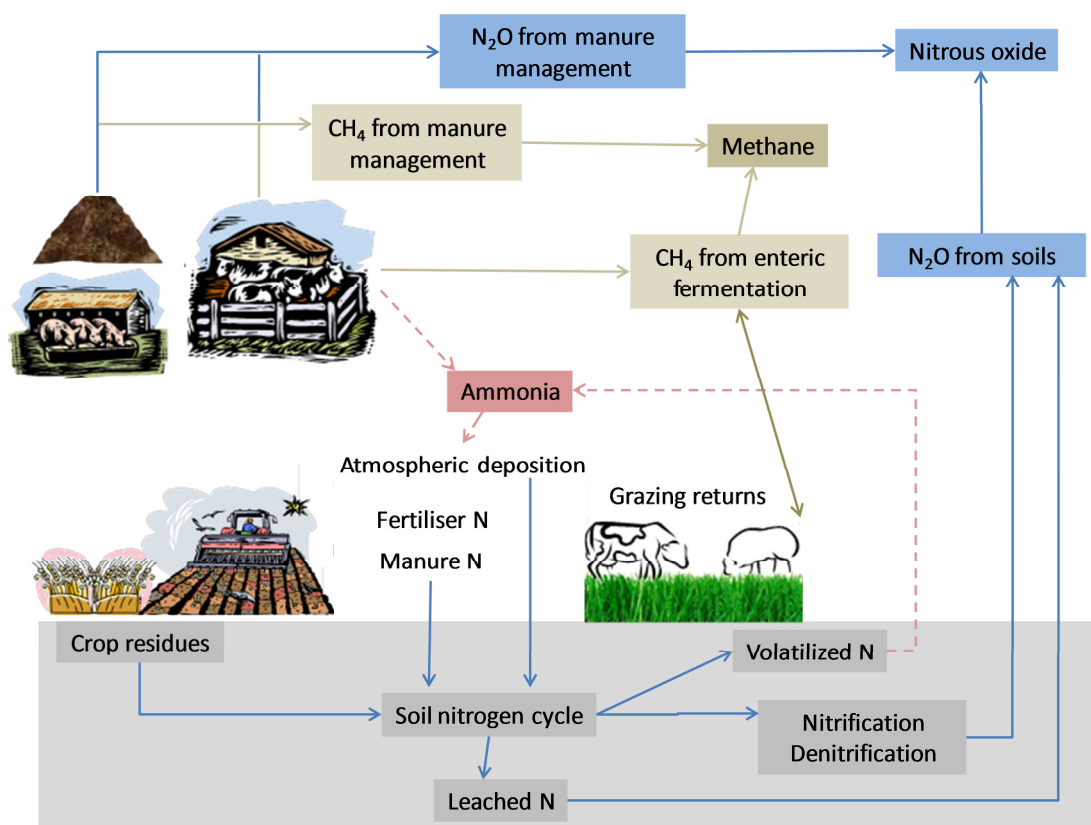


Figure 3. Sources of GHG and  $\text{NH}_3$  emissions on farm  
(Figure courtesy of T. Misselbrook, Rothamsted Research, personal communication)

The potential synergies and trade-offs affect our mitigation efforts and need to be taken into account when optimising abatement activities. Focusing on a single pollutant can lead to under- or overestimating the total benefit of pollution control, and thus to suboptimal mitigation effort (Nemet *et al.* 2010). Economic efficiency is an important consideration in environmental policy formulation. Regulatory interventions should aim to reduce pollution at least cost, or at least in ways where costs are demonstrably outweighed by benefits; the latter quantified in terms of avoided damages. This criterion involves a comparison of private and what economists term social costs, which are essentially the wider environmental costs and benefits of pollution control.

Most decisions in livestock systems design, animal feeding, manure management and crop fertilisation are likely to affect more than one of the processes mentioned above. To support policy decisions, integrated assessment of the mitigation of  $\text{NH}_3$  and GHGs is needed. This chapter reviews current knowledge on the positive and negative co-effects of  $\text{NH}_3$  abatement measures in agriculture, focusing on the GHGs  $\text{N}_2\text{O}$  and  $\text{CH}_4$ .

The next section provides background on agricultural emissions of NH<sub>3</sub> and GHGs, sections 2.4 – 2.7 discuss the likely co-effects of NH<sub>3</sub> MMs in various areas of farm management, and conclusions drawn in Section 2.8.

This review looks at these pollutants as they emerge from the grounds of the farms. Nevertheless, some important implications on emissions beyond the farm gate, for example GHG emissions from fertiliser production, are mentioned. The focus is on temperate farming systems in Europe, though the experimental evidence reviewed goes beyond Europe.

## 2.3 Ammonia and greenhouse gas emissions in agriculture

### 2.3.1 Ammonia

NH<sub>3</sub> contributes to acidification and eutrophication in marine and terrestrial ecosystems, and it also has detrimental effects on human health (Smart *et al.* 2011). A small part of the NH<sub>3</sub> released into the environment is converted into N<sub>2</sub>O, which is a powerful GHG. Agriculture is responsible for 94% of NH<sub>3</sub> emissions in the EU-27 countries, the remainder coming from road transport, waste and industrial processes (European Environment Agency 2013) (Table 3).

*Table 3. European N<sub>2</sub>O, CH<sub>4</sub>, CO<sub>2</sub> and NH<sub>3</sub> emissions and the contribution of main agricultural activities to the agricultural emissions (EU-27, 2011)*

	N <sub>2</sub> O	CH <sub>4</sub>	CO <sub>2</sub>	NH <sub>3</sub>
Emissions	(Mt CO <sub>2</sub> e)	(Mt CO <sub>2</sub> e)	(Mt CO <sub>2</sub> )	(kt NH <sub>3</sub> )
Total emissions	337	397	3,747	3,635
Agricultural emissions	275	197	0 <sup>1</sup>	3,394
<b>Contribution to agricultural emissions</b>				
Enteric fermentation	0%	74%	n/a	0%
Manure management	11%	24%	n/a	74%
Rice cultivation	0%	1%	n/a	0%
Agricultural soils	89%	0%	n/a	25%
Field burning	0%	0%	n/a	0%

<sup>1</sup> CO<sub>2</sub> emissions from agriculture are accounted for under other categories

Source: (European Environment Agency 2013, European Environment Agency 2014)

NH<sub>3</sub> originates both from livestock and arable farming. N not retained by livestock is excreted in faeces and urine excreta; the former mainly contain organic N compounds, while

the N in urine is mainly non-protein N (mostly urea) (Monteny and Erisman 1998). Birds excrete uric acid which is readily hydrolysed to urea (Webb 2001). The urea can be quickly hydrolysed into  $\text{NH}_3$  by the enzyme urease, which can be found in the faeces, on fouled surfaces and in soil. On the other hand, the protein-N of faeces first has to go through the slow process of mineralisation to become part of the total ammoniacal nitrogen (TAN) pool (i.e.  $\text{NH}_3$  and ammonium,  $\text{NH}_4^+$ ), therefore  $\text{NH}_3$  volatilisation is much lower from faeces (Bussink and Oenema 1998). All in all, the N content of the excreta is partially lost as  $\text{NH}_3$  from the livestock houses and manure stores and from the fields either after being deposited during grazing or having been applied to soils as a fertiliser. As for cropping activities, inorganic N fertilisers are also sources of  $\text{NH}_3$  emissions, but a great difference exist according to the type of fertiliser and the application method (Hutchings *et al.* 2001, Misselbrook *et al.* 2000).

Various physical and biological factors have an effect on what proportion of the N in livestock excreta and in inorganic fertilisers are being lost as  $\text{NH}_3$ .  $\text{NH}_3$  emissions are positively correlated with pH, temperature and air velocity and also increase with higher urease concentration (Bouwmeester and Vlek 1981, Carmona *et al.* 1990, Sommer *et al.* 1991). At the same time the  $\text{NH}_3$  can be converted into other N compounds by processes like microbial immobilisation, assimilation by plants, and nitrification (Rennenberg *et al.* 2009), reducing the TAN content and thus  $\text{NH}_3$  emissions.

### 2.3.2 Nitrous oxide

$\text{N}_2\text{O}$  is a potent greenhouse and, at the same time, the most important ozone depleting substance (Ravishankara *et al.* 2009). Primary human-related sources of this gas are agriculture, and, to a lesser extent, combustion and industrial processes, the former contributing 50% of the total  $\text{N}_2\text{O}$  emissions in Europe (European Environment Agency 2014), and 75% of the global total (EPA 2012). Most of the agricultural emissions are produced in soils, with a lesser amount generated during manure management (Table 3): the N added to soils (e.g. inorganic and organic fertilisation, crop residues, atmospheric deposition, livestock excreta on pastures) and excreted by livestock in animal houses are the main sources of  $\text{N}_2\text{O}$ . Additionally, soluble nitrogen compounds leached into water bodies and gaseous  $\text{NH}_3$  emissions can also be converted into  $\text{N}_2\text{O}$ .

The two main processes of  $\text{N}_2\text{O}$  generation are nitrification and denitrification (Figure 4). In nitrification, in aerobic conditions  $\text{NH}_4^+$  is transformed into nitrite and then into  $\text{NO}_3^-$ , and, particularly in low oxygen concentration,  $\text{N}_2\text{O}$  is emitted (Bremner and Blackmer 1978). Subsequently, denitrifying bacteria convert  $\text{NO}_3^-$  into  $\text{N}_2$  gas in anoxic conditions. However, if the concentration of molecular oxygen increases, the formation of  $\text{N}_2\text{O}$  rather than  $\text{N}_2$  is promoted through incomplete denitrification (Firestone *et al.* 1980). As the nitrifying and denitrifying bacteria require different oxygenation level, aerobic and anaerobic pockets being in close proximity to each other favour very high  $\text{N}_2\text{O}$  emissions. The production of  $\text{N}_2\text{O}$  depends on the  $\text{NH}_4^+$  and other N compounds' concentration in the environment (which are all related to soil properties and manure composition) and on temperature: warm conditions promote bacterial growth, but temperatures above approx. 50 °C inhibit it, because nitrifying and denitrifying bacteria are not thermophilic (Sommer and Moller 2000).

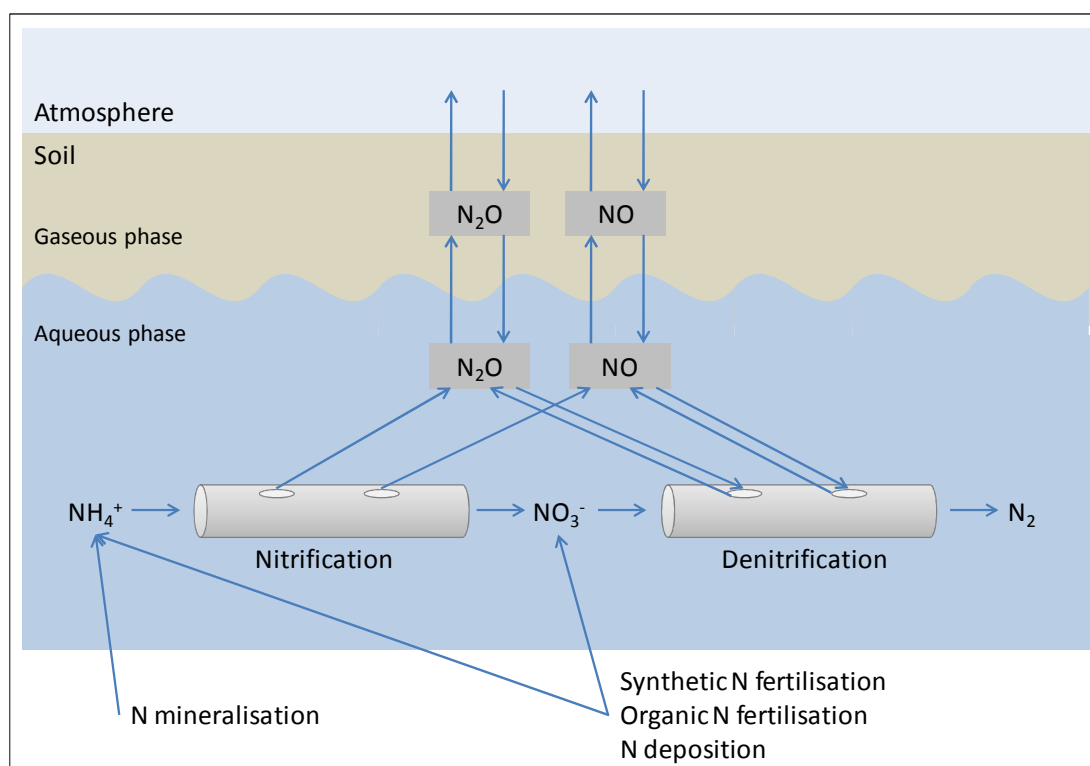


Figure 4. 'Hole-in-the-pipe' model  
Model of the regulation of trace-gas production and consumption by nitrification and denitrification (adopted from Bouwman 1998)

Agricultural soils are important sources of  $\text{N}_2\text{O}$ ; on average 1% of N added to the soils escapes to the air directly as  $\text{N}_2\text{O}$  (IPCC 2006). These emissions are enhanced during wet and warm conditions. Livestock operations generate  $\text{N}_2\text{O}$  emissions mainly through solid manure storage and in livestock bedding, but the surface layers of slurry can also emit  $\text{N}_2\text{O}$

(Chadwick *et al.* 2011). Additionally, the  $\text{NH}_3$  emitted by agricultural activities is an indirect source of  $\text{N}_2\text{O}$ .

### 2.3.3 Methane

Globally and in the EU-27 approximately half of anthropogenic  $\text{CH}_4$  emissions originate from agriculture, dominated by enteric fermentation; while the other half mainly arises from gas drilling, coal mining and landfill (European Environment Agency 2014). In 2010 global agricultural  $\text{CH}_4$  emissions were dominated by enteric fermentation (62%), followed by rice cultivation (17%) and livestock waste (7%) (EPA 2012). The pattern in Europe is similar, with the notable difference of emissions from rice cultivation being marginal (Table 3).

$\text{CH}_4$  is produced by anaerobic respiration of methanogen microorganisms. This process happens when the breakdown of organic material takes place in the lack of oxygen and no other electron acceptors are present but small organic compounds and  $\text{CO}_2$ . The release of  $\text{CH}_4$  intensifies with higher temperatures (Khan *et al.* 1997), even above  $50^\circ\text{C}$ , as many methanogens are thermophil microorganisms (Sommer and Moller 2000). Eventually the emitted  $\text{CH}_4$  is oxidised back to  $\text{CO}_2$  in the atmosphere.

Environments favouring methanogenesis occur in the digestive system of animals, in manure stores and in anoxic soils, like wetlands and rice paddies. In animals methanogenesis happens during bacterial fermentation of feedstuff in the rumen of cattle, sheep and other ruminants, and also occurs, to a lesser extent, in the large intestine of all livestock. Manure management is also responsible for  $\text{CH}_4$  emissions, where these emissions originate from the anaerobic decomposition of livestock bedding and manure, especially in liquid manure stores. The manure composition (especially the proportion of volatile solids) and the length of the anaerobic storage period are important factors in determining the  $\text{CH}_4$  emissions. While the  $\text{CH}_4$  emissions of ruminants are mainly produced in the rumen, those from pigs and poultry are mostly manure-born (Table 4).

*Table 4. Contribution of the different livestock species to CH<sub>4</sub> emissions from enteric fermentation and manure management (EU-28, 2011)*

	Enteric fermentation	Manure management
Cattle	82%	49%
Sheep	12%	1%
Pigs	3%	44%
Poultry	0%	4%

*Source: (European Environment Agency 2014)*

### 2.3.4 Carbon dioxide

Although considering all anthropogenic GHG emissions CO<sub>2</sub> contributes the most to global warming, its importance in agriculture is tertiary to N<sub>2</sub>O and CH<sub>4</sub>, its main sources being land use, land use change and fossil fuel combustion. Agricultural land use activities, particularly changes in the land use, e.g. from cropland to grassland or vice versa, result in a positive or negative change in the soil carbon (C) stocks. The former process removes CO<sub>2</sub> from the atmosphere (C sequestration), the latter releases CO<sub>2</sub>, for example cropland and grassland related land use and land use change added 78 Mt CO<sub>2</sub>e emissions to the EU-27 inventory in 2011 (European Environment Agency 2014). In the same year fossil fuel combustion (agriculture together with forestry and fisheries) contributed with a further 75 Mt CO<sub>2</sub>e to the total emissions (European Environment Agency 2014).

## 2.4 Dietary options

Animal nutrition has considerable effects on NH<sub>3</sub> and GHG emissions, both directly and indirectly. Optimal feed composition and additional factors (e.g. water and feed availability, temperature in the stalls) facilitates higher energy and protein use efficiency and improves animal health (Roche 2006, VandeHaar and St-Pierre 2006), and thus reduces waste directly at the animal level. Good feeding practice can boost the physical efficiency at the farm level as well, reducing waste indirectly. For example, dietary factors play an important role in both the age of first breeding and in the fertility of dairy cattle (VandeHaar and St-Pierre 2006), impacting on the length of unproductive periods and on the need for replacement heifers in the herd. This section focuses on dietary measures targeting N intake and briefly presents two additional options relevant for piggeries. These MMs impact on the whole N cascade,



and thus they effect direct and indirect N<sub>2</sub>O emissions, and in some cases they also effect enteric and manure CH<sub>4</sub> emissions.

Removal of the excess N from the feed is a widely proposed feeding measure to control NH<sub>3</sub> emissions through the reduction of N excreta. Though significant progress have been made in some European regions in this respect (Dalgaard *et al.* 2012, Groot *et al.* 2006), farmers still often feed livestock with excess protein in order to avoid the risk of reduced production due to inadequate N intake (Aarnink and Verstegen 2007, Aberystwyth University 2010). One survey in the USA showed that farmers, on average, fed 6.6% more N than was recommended by the National Research Council, resulting in an increase of 16% and 2.7% in urinary N and faecal N excretion, respectively (Jonker *et al.* 2002). Such excess in N inputs can be avoided without productivity loss (Hristov *et al.* 2011, Rotz 2004).

A range of practical solutions have been suggested to achieve the aforementioned reduction in nitrogen intake, which translates into a reduction in the protein content of the diet with a parallel increase of non-protein substrates (often carbohydrates). Monogastric animals are fed with compound feeds, where the crude protein (CP) content reduction can be achieved by replacing part of the high N content feed components with components rich in energy or fibre. In the case of ruminants there is scope to alter the ratio of forage versus concentrate feeds, the CP content of the concentrates by altering their composition (e.g. more components rich in starch) and the CP content of the forage (for example by providing starch-rich maize silage, changing the grass varieties or reducing the grass fertilisation rate). Where the low-protein diet is limited in essential amino acids (AAs) then supplementing these to balance AA composition might be needed to maintain production levels (Aarnink and Verstegen 2007, VandeHaar and St-Pierre 2006). For ruminants the AAs must be in rumen-undegradable form to go under enzymatic digestion and absorbed by the animal itself rather than its microorganisms (Broderick *et al.* 2008).

A reduced N intake translates into reduced N excretion; there is a linear relationship between dietary CP content and N excretion in dairy and beef cattle (Hristov *et al.* 2011, Waldrip *et al.* 2013). The drop in the N excretion is mostly due to a decrease in the urinary N, while the faecal N remains relatively constant (Bussink and Oenema 1998). As NH<sub>3</sub> volatilisation is much higher from the urine than from the faeces, the saving in NH<sub>3</sub> emissions can be proportionally higher than the savings in the N excretion (Rotz 2004). Hristov *et al.* (2011) provides a summary of experiments reporting 28-50% reduction in NH<sub>3</sub> emissions from cattle manure storage and parallel reduction in NH<sub>3</sub> emissions after soil application of the

manure when the CP content of the diet was altered from high level (between 15.4-17.5%) to low level (between 12.5-14.8%). The relationship between N intake and excretion and NH<sub>3</sub> volatilisation is similar for pigs and poultry to that of cattle. Reducing the CP content of the diet while administering essential AAs can reduce N excretion in pigs and poultry and hence leads to a reduction in NH<sub>3</sub> emissions (Rotz 2004). An additional effect of the reduced protein intake in pigs is that the manure becomes more acidic, further decreasing NH<sub>3</sub> volatilisation (Canh *et al.* 1998a).

The GHG effects of the reduced N intake are multiple. The reduced NH<sub>3</sub> emissions imply lower indirect N<sub>2</sub>O emissions from ammonia volatilisation, and the reduced N excretion is expected to translate into reduced direct N<sub>2</sub>O emissions from manure storage and application, for example the IPCC Tier 2 calculations assume a linear relationship between direct manure storage N<sub>2</sub>O emissions and crude protein intake (IPCC 2006). However, experimental evidence is not conclusive in this respect (see below) (Philippe and Nicks 2015). Furthermore, the changes in the feed composition can alter the CH<sub>4</sub> emissions both from manure storage and from enteric fermentation. Regarding the latter, if in the low protein feed the energy replaced comes from fibre or sugars, more enteric CH<sub>4</sub> is likely to be produced, whereby if it comes from starch or fat, CH<sub>4</sub> emissions can be reduced (Dijkstra *et al.* 2011).

Looking at GHG effects of reduced CP content in the diet, Philippe *et al.* (2006) found a net increase of 19% in GHG emissions from the buildings of pigs kept on deep litter. The two-phase diet CP content was 18.1% and 17.5%, respectively, for growers and finishers in the high protein group and was 15.5% and 14.0%, respectively, for the two growth stages in the low protein group, (the latter diets were supplemented with AAs). While NH<sub>3</sub> emissions from the low protein group were significantly lower than from the high protein group, N<sub>2</sub>O emissions doubled for the former group, and this was only partially offset by the reduced CH<sub>4</sub> emissions. Conflicting results exist on the consequence of low protein diet on GHG emissions from manure storage: Külling *et al.* (2001) reported increased CH<sub>4</sub> and reduced N<sub>2</sub>O emissions with zero net GHG effect for dairy manure, Velthof *et al.* (2005) found reduced CH<sub>4</sub> emissions from pig manure, whereas there was no statistical difference for either GHGs in a third experiment (Lee *et al.* 2012) on dairy manure. Kreuzer and Hindrichsen (2006) imply that the C:N ratio of the manure is a more important factor in the CH<sub>4</sub> emissions from manure storage than the N content, with a low C:N ration resulting in higher CH<sub>4</sub> emissions. The complex effect of reduced CP content on GHG emissions from

manure application directly affected by further factors such as volatile fatty acid content of the manure (Sommer *et al.* 2004), the type of manure management, soil characteristics and weather conditions. The direction of change in N<sub>2</sub>O emissions in a soil incubation study varied with soil type after application of pig manure (Velthof *et al.* 2005), while no statistical difference was observed between GHG emissions from manure application following feeding dairy animals with high and low protein diets (Lee *et al.* 2012); Misselbrook *et al.* (1998) found no change in N<sub>2</sub>O emissions, although CH<sub>4</sub> emissions were reduced with lower CP content (however, CH<sub>4</sub> emissions from manure application are marginal).

The CP content of the diet is often reduced with a correspondent increase in the starch content. This has a positive side-effect on GHG emissions, more starch also leads to lower enteric CH<sub>4</sub> emissions in ruminants (Aberystwyth University 2010, Mc Geough *et al.* 2010a, Mc Geough *et al.* 2010b, Moe and Tyrrell 1979), since CH<sub>4</sub> production originates mainly from the by-products of structural polysaccharide (e.g. cellulose) fermentation (Ellis *et al.* 2008). It should also be noted that too much starch is detrimental to the animal health as it causes rumen acidosis (Owens *et al.* 1998), and feeding high levels of concentrates diminishes the main environmental benefit of cattle: converting structural polysaccharide (not only grass, but fibrous by-products, like almond hulls, citrus pulp) into high-quality protein for human use (Oltjen and Beckett 1996, VandeHaar and St-Pierre 2006). Additionally, the net GHG saving achievable with this method is questionable, as the soil C content of land under arable cultivation (i.e. silage maize) is lower than that of grasslands, and such a change in land use results in CO<sub>2</sub> emissions from soil (Beauchemin *et al.* 2010, Vellinga and Hoving 2011).

If the CP content is partially replaced by dietary fats in ruminant diets, enteric CH<sub>4</sub> emissions are reduced, partly due to a suppression of some of the rumen microflora and to a lower extent due to unsaturated fatty acids acting as hydrogen sinks in the rumen (Johnson and Johnson 1995, Martin *et al.* 2010). The savings in enteric CH<sub>4</sub> emissions is proportional to the amount of fat in the diet (Beauchemin *et al.* 2008, Eugene *et al.* 2008, Grainger and Beauchemin 2011) and can be increased up to 5-6% without adverse nutritional effects. According to Hristov *et al.* (2013) and Martin *et al.* (2010), the question of persistence of the mitigation effect has not been adequately addressed yet: some studies do report long-term effects, but data are inconsistent. In addition, two mechanisms might (partially) off-set the savings in enteric CH<sub>4</sub> emissions: potential increases in manure storage CH<sub>4</sub> emissions (Kulling *et al.* 2002) and in emissions related to the production of feedstuff, especially if

they induce a land use change deteriorating soil C stocks, for example via an increase in palm oil plantations.

As Peyraud and Astigarraga (1998) summarise in a review, with decreasing amount of N fertiliser applied, the protein content of the grass substantially decreases and the water soluble carbohydrate content increases. Livestock feeding on such grass excrete markedly reduced urinal N, thus related  $\text{NH}_3$  emissions are lower. With this method, fertiliser related  $\text{NH}_3$  and  $\text{N}_2\text{O}$  emission savings are also achieved through reduced fertiliser use per land area, though this benefit can be outweighed by the lower grass yield and thus the potential reduction in soil carbon stocks, if maintaining livestock production leads to a conversion of woodlands or wetlands into grazing land. Similarly, using high sugar content grass varieties can also improve N efficiency of cattle by increasing the capture of N into microbial protein, and thereby increasing milk protein outputs and at the same time reducing urinary N excretion (Moorby *et al.* 2006). However, lowering the N fertilisation of the grass affects ruminants' enteric  $\text{CH}_4$  emissions, though research in this respect is so far inconclusive (Dijkstra *et al.* 2011). Similarly, contrasting results exist on the enteric  $\text{CH}_4$  effects of the high sugar content grasses; a modelling exercise by Ellis *et al.* (2011) presented variable results on  $\text{CH}_4$  emissions, depending on the concurrent changes in the diet and the measurement unit, i.e. whether results were expressed as percentage of gross energy intake or grams per kg of milk.

As discussed earlier, farmers often perceive a high risk of a reduction in productivity in response to lower protein intake. Falling production results in both financial losses to the farmers and a possible increase in pollutant load per production unit (Weiske 2005). An increasing reliability of feed recommendation systems should help to provide the confidence to farmers in better diet formulation (Cuttle *et al.* 2004). In addition, stricter quality control of feed materials could also help to balance nutrients (Nahm 2002). However, as St-Pierre and Thraen (1999) showed, there might be a discrepancy between the maximum physical efficiency and the maximum economic efficiency, causing overfeeding of the animals, though this discrepancy varies not only with livestock and crop varieties but also with the changes in the relative price level of N inputs and products.

Beyond lowering the N intake of the animals, two more dietary MMs aiming to reduce  $\text{NH}_3$  emissions from pig farms are discussed here. First, providing a higher non-starch polysaccharide content diet (e.g. sugar beet pulp) with a constant CP concentration decreases both the urinary-N/faecal-N ratio and the faeces pH (Canh *et al.* 1997), potentially lowering

NH<sub>3</sub> volatilisation. But with more fermentable polysaccharides, volatile fatty acid concentration increases, and thus increases CH<sub>4</sub> emissions (Velthof *et al.* 2005). A further economic and environmental disadvantage of this MM is that using higher digestibility raw materials decreases the use of low-cost by-products (Edwards *et al.* 2002). Second, the urine pH can be lowered by replacing calcium carbonate with calcium sulphate in the pig diet, reducing NH<sub>3</sub> volatilisation (Canh *et al.* 1998b). At the same time the lower ileal pH might reduce CH<sub>4</sub> emissions from liquid manure stores (Kim *et al.* 2004). Indeed, Velthof *et al.* (2005) found that this technique reduced both CH<sub>4</sub> and NH<sub>3</sub> emissions from anaerobic storage, but N<sub>2</sub>O emissions from soil incubation were variable, depending on the soil type.

## 2.5 Livestock housing

During periods spent in houses animals excrete the urine and faeces onto hard surfaces, onto the bedding or into slurry pits, where a substantial part of the TAN content volatilises – housing emissions constitute for ¼ of agricultural NH<sub>3</sub> emissions in the UK (Misselbrook *et al.* 2012). A range of housing characteristics impacts on the gaseous emissions, including the manure handling system (liquid or solid), the housing design (e.g. ventilation type and airflows, floor surface, inside or outside manure pit), management decisions (e.g. frequency of manure removal, manure additives) and climatic conditions. Numerous management and technical MMs exist both in slurry-based and litter-based systems to reduce NH<sub>3</sub> emissions, and in many of them affect GHG emissions as well. Below five MMs are summarised.

In litter-based cattle and pig systems increasing the amount of straw has a positive effect on both NH<sub>3</sub> and GHGs. Adding extra straw bedding (25-50% above typical practice), targeting especially the wetter and more fouled areas is effective on in-house and manure storage related NH<sub>3</sub> emissions (Gillespie *et al.* 2009, IGER 2005). More straw reduces air-flow and consequently volatilisation while at the same time the higher C:N ratio enhances immobilisation of NH<sub>4</sub>-N, reducing NH<sub>3</sub> emissions considerably (Dewes 1996). The additional straw efficiently reduces N<sub>2</sub>O emissions (Sommer and Moller 2000, Yamulki 2006) and might reduce or increase CH<sub>4</sub> emissions, as the better aeration inhibits anaerobic methanogens, but the additional carbohydrates provide extra substrate for methanogens (Philippe and Nicks 2015).

Converting fully-slatted floors in pig houses to partly slatted floors can reduce the fouled surface area by encouraging the pigs to dung over the slatted area, and reduces air exchange

between the pit and the house – thus reducing  $\text{NH}_3$  emissions (Aarnink *et al.* 1997).

Concerning both  $\text{CH}_4$  and  $\text{N}_2\text{O}$  emissions contradictory results have been published, with no consensus yet on the effects (Philippe and Nicks 2015).

In the straw-flow systems in pig houses (Bruce 1990) the straw is added at the top of a sloped lying area and it travels down the slope towards an excretion area, where it mixes with the dung. The manure then leaves the excretion area either into an underneath pit or onto a scraped passage, depending on the actual design. According to the amount of added straw the produced manure is either slurry or solid manure. Amon *et al.* (2007) experienced reduced in-house  $\text{NH}_3$ ,  $\text{CH}_4$  and  $\text{N}_2\text{O}$  emissions from this system compared to fully slatted floor systems, while Philippe *et al.* (2007) found only the  $\text{CH}_4$  emissions to be lower from straw flow system compared to houses with fully slatted floors, with  $\text{N}_2\text{O}$  emissions being at the same level and  $\text{NH}_3$  emission being 2.5 times higher.

Since  $\text{NH}_3$  emissions are positively correlated with temperature, frequent removal of manure from pig and cattle in-house storage to outdoor storage reduces  $\text{NH}_3$  emissions, given the temperature is lower outside than inside (Hartung and Phillips 1994). As  $\text{CH}_4$  emissions increase with temperature, they can also be reduced with this MM by 10-19% from both liquid and solid systems, and though  $\text{N}_2\text{O}$  emissions might increase, they stay negligible compared to  $\text{CH}_4$  (Philippe and Nicks 2015). Pit flushing and scraping in piggeries can concurrently reduce  $\text{CH}_4$  and  $\text{N}_2\text{O}$  emissions, the more frequent the flushing or scraping, the higher GHG savings to achieve. Regarding poultry, keeping the poultry manure dry or drying it on manure belts saves  $\text{NH}_3$  emissions, but a study comparing laying hen housing systems with aerated deep pit manure storage and with forced drying manure belt removal found higher  $\text{CH}_4$  emissions from the removal system (Fabbri *et al.* 2007).

Finally, installing air scrubbers or biofilters to remove  $\text{NH}_3$  from animal houses is a very efficient end-of-pipe technology used for mechanically ventilated houses (Melse *et al.* 2009). Air scrubbers work on a chemical basis:  $\text{NH}_3$  is captured as  $\text{NH}_4^+$  salt in an acidic solution (mostly sulphuric acid). Biofilters use microorganisms to convert  $\text{NH}_3$  into  $\text{NO}_3^-$ . In both cases, the discharge water can be used as fertiliser. Slightly increased  $\text{CO}_2$  emissions arise from the increased energy usage if mechanical ventilation is only installed for the air filtering purposes. Furthermore, in biofilters there is a risk of increased  $\text{N}_2\text{O}$  emissions due to the nitrification process in the filter (Melse and van der Werf 2005); one study found that 20% of the biofilter's N content was released as  $\text{N}_2\text{O}$  (Maia *et al.* 2012).

## 2.6 Manure storage

Animal excreta are a crucial source of gaseous emissions from agriculture, either as deposited on pastures during grazing, or collected, stored, and subsequently applied as fertiliser. The emission profile of various manure handling systems are markedly different: liquid systems are generally an important source of  $\text{NH}_3$  and  $\text{CH}_4$  emissions, while solid systems emitting more  $\text{N}_2\text{O}$  and less ammonia and  $\text{CH}_4$ . In both cases several factors play important roles in the emissions, like the initial composition of manure (e.g. N content, TAN content, C:N ratio, water content, dry matter (DM) content, volatile solids (VS) content), the manure store characteristics (e.g. covered or not), the management decisions (e.g. length of in-house storage and outside storage, aeration level, additives) and the environmental variables (e.g. temperature, rainfall) all affect the gaseous emissions and leaching from manure (Monteny *et al.* 2006, Sommer *et al.* 2004). Solutions favouring lower  $\text{NH}_3$  emissions in both types of systems are very likely to have synergies or trade-offs with the GHGs.

### 2.6.1 Liquid manure

In conventional liquid manure storage, where the slurry is not aerated, the anaerobic environment does allow denitrification happening only at a very low rate, close to the surface, producing only small amount of  $\text{N}_2\text{O}$  and holding back subsequent denitrification which would also be a source of  $\text{N}_2\text{O}$  (Sommer *et al.* 2000, Zhang *et al.* 2005). On the other hand, the anaerobic environment is ideal for methanogen microorganisms, making slurry stores an important source of  $\text{CH}_4$  emissions. The GHG effects of covering slurry stores, separating the slurry into solid and liquid fractions and slurry acidification is presented here.

Covering slurry stores substantially reduces  $\text{NH}_3$  emissions (VanderZaag *et al.* 2008). As a result, the TAN content of the slurry increases, and it will be susceptible to elevated emission levels after having been spread on the soil, unless low  $\text{NH}_3$ -emission spreading techniques (see section 2.7) are implemented. The effects of covering slurry stores on GHGs are less explored than the consequences on  $\text{NH}_3$ , and the results are highly variable and inconclusive (VanderZaag *et al.* 2008), as presented below via selected examples from the wide range of covering options.

Floating covers can be made of organic (e.g. straw, vegetable oil), inorganic (expanded clay) or synthetic materials. If manure properties allow and the slurry is not agitated, natural crust can develop on the surface, especially on cattle slurry (Chadwick *et al.* 2011). The crust development can be artificially enhanced by covering the surface with straw or Leca (expanded clay) pebbles. Though greatly reducing NH<sub>3</sub> emissions, crust and straw cover provides suitable conditions to nitrifying bacteria and thus provoke a dramatic increase in N<sub>2</sub>O emissions, especially in dry weather (Berg *et al.* 2006, Sommer *et al.* 2000). At the same time these surface layers can be colonised by methanotroph bacteria, oxidising part of the CH<sub>4</sub> to CO<sub>2</sub> (Petersen and Ambus 2006, Petersen *et al.* 2005): Sommer *et al.* (2000) observed a similar 28% reduction with straw, leca and crust cover and VanderZaag (2009) also noted 24-28% savings in CH<sub>4</sub> emissions with straw cover. However, a reduction in the CH<sub>4</sub> emissions is not always observed (Berg *et al.* 2006, Hudson *et al.* 2006, Petersen *et al.* 2013). On the other hand, permeable synthetic cover though reduces N<sub>2</sub>O emissions, the overall GHG emissions are not affected substantially due to no significant effect on CH<sub>4</sub> emissions (VanderZaag *et al.* 2010).

Rigid covers (e.g. wooden or concrete lids or tent structures) may also be promising for CH<sub>4</sub> emission reductions: Clemens *et al.* (2006) reported 14-16% savings in CH<sub>4</sub> emissions from crusted cattle slurry if covered with wooden lid, Amon *et al.* (2006) found that CH<sub>4</sub> emissions were 18% lower from lid-covered than from straw-covered cattle slurry, though CH<sub>4</sub> emissions might increase as well (Silsoe Research Institute 2000). The effect of solid covers on N<sub>2</sub>O emissions is more variable, some research showing benefits others disadvantages (Amon *et al.* 2006, Clemens *et al.* 2006, Petersen *et al.* 2009, Silsoe Research Institute 2000).

Finally, impermeable floating or rigid covers can be equipped with gas pipes and pumping systems to collect the gas produced. In such systems most of the CH<sub>4</sub> is captured and converted to CO<sub>2</sub> either by direct flaring, reducing the GWP substantially, or by purification and use in electricity or heat generation, providing further GHG benefits by replacing non-renewable energy sources (Petersen and Miller 2006).

The mechanical or chemical separation of slurry produces a solid and a liquid fraction with markedly different gaseous emission patterns; the solid fraction akin to untreated solid manure while the liquid fraction is similar to slurry. So far the results show contrasting effects on NH<sub>3</sub> and GHG emissions: the former is often higher from the separated slurry than from the unseparated slurry (Amon *et al.* 2006, Dinuccio *et al.* 2008, Dinuccio *et al.* 2011,



Fangueiro *et al.* 2008), while the overall CH<sub>4</sub> and N<sub>2</sub>O emission is reduced by separation by as much as 26-37% (Amon *et al.* 2006, Fangueiro *et al.* 2008), though Dinuccio *et al.* (2008) observed higher GHG emissions as well. The GHG is attributable to a drop in CH<sub>4</sub> emissions, usually counter-balancing the – sometimes considerably – increased N<sub>2</sub>O emissions. Regarding emissions from the application of separated and not separated slurries, Amon *et al.* (2006) found higher overall GHG emissions from the separated slurry, but as field application GHG emissions were only 1.3% of the total GHG emissions, the increase only slightly reduced the net GHG benefits.

Slurry acidification can reduce NH<sub>3</sub> emissions from housing, storage and application by 10-60% (Kai *et al.* 2008, Monteny and Erisman 1998), though Berg *et al.* (2006) reported increased ammonia emissions from acidified slurry covered by perlite or Leca. They also found 43-76% less CH<sub>4</sub> emissions from the acidified tanks, and an earlier research showed that pH below 5.0 substantially reduced CH<sub>4</sub> emissions, while pH < 4.5 almost completely mitigated them (Berg and Hornig 1997). A recent paper investigated the effects of acidification on NH<sub>3</sub> and CH<sub>4</sub> emissions, and found 93-98% and 67-87% reductions, respectively (Petersen *et al.* 2012). When applied on land, acidification delays nitrification and N<sub>2</sub>O formation, and total emissions of N<sub>2</sub>O might also be reduced if the slurry is also separated (Fangueiro *et al.* 2010).

## 2.6.2 Solid manure

Traditionally, solid manure was composted, i.e. the manure was aerated by turning the heap several times during the storage period. As composting progresses, the physical and biological circumstances and microbial communities change substantially, leading to a temporal pattern in the gaseous substances generated. The compost heap also has a significant spatial heterogeneity, supplemented with a prominent temperature and O<sub>2</sub> gradient from the surface to the centre. Furthermore, climatic conditions modify the surface layer of the heap, altering mainly the N<sub>2</sub>O emissions (Petersen *et al.* 1998).

The first phase of composting is characterised by high microbial activity, quick decomposition of easily degradable substances, high CO<sub>2</sub> emissions, intensive heat production, depletion of acidic components, with very low CH<sub>4</sub> emissions and decreasing N<sub>2</sub>O emissions (Hellmann *et al.* 1997). The second phase (thermophilic phase) is a high temperature phase, with quickly declining CO<sub>2</sub> emissions but high CH<sub>4</sub> emissions from the

centre of the heap. N<sub>2</sub>O emissions are restricted to the surface in this phase first because the nitrifying and denitrifying bacteria are not thermophilic (Sommer and Moller 2000), and secondly because of the anaerobic environment in the centre of the heap. Close to the surface anaerobic and aerobic pockets close to each other allow for the NH<sub>4</sub><sup>+</sup> to be nitrified and the NO<sub>3</sub><sup>-</sup> to be denitrified (Hansen *et al.* 2006, Hellmann *et al.* 1997). In the third phase (curing phase) the CH<sub>4</sub> emissions are decreasing and the N<sub>2</sub>O emissions are increasing due to the lower temperature, whereas the CO<sub>2</sub> emissions remain low (Hellmann *et al.* 1997).

The physical and biological circumstances of solid manure storage can be altered by different practices, like waterproof cover, anaerobic storage (airtight cover), compression at the beginning of storage, cut and mix before storage, aeration, or adding extra straw to the manure. Consequences of these MMs on the GHG emissions are discussed below.

By compaction or airtight covering an increased proportion of the solid manure heap becomes anaerobic, reducing NH<sub>3</sub> emissions by 19-98% (Amon *et al.* 2001, Amon *et al.* 1997, Chadwick 2005, Kirchmann and Witter 1989), though with variable effects on GHGs. Chadwick (2005) reported about inconclusive GHG effects of simultaneous compaction and covering of the manure heap. N<sub>2</sub>O emission changes ranged from -71% to 19-fold increase, the effect on CH<sub>4</sub> emissions were from -78% to +139%. Sommer's (2001) results for porous covering and compacting the heaps showed that these MMs increased both GHG emissions, porous covering resulting in moderate increase, while compacting leading to 1.5 and 5.5-fold increase in N<sub>2</sub>O and CH<sub>4</sub> emissions, respectively. Amon *et al.* (2001) compared anaerobically stacked heap with one composted, and found higher GHG emissions from the stacked heap (+7% in winter, and +347% in summer). On the other hand, results from Hansen *et al.* (2006) suggest that the gaseous emissions from the separated solids of anaerobically digested pig slurry can simultaneously be reduced: airtight covering decreased the emissions of NH<sub>3</sub>, N<sub>2</sub>O, CH<sub>4</sub> and CO<sub>2</sub> by 12%, 99%, 88% and 93%, respectively. The authors explained the reduction in CH<sub>4</sub> emissions despite the anaerobic conditions by the lower temperature of the heap, which is not favourable to methanogens.

Adding extra straw to the farmyard manure reduces the density, increases the C:N ratio, increases the porosity of the manure and enhances the airflow within the heap (affecting both the oxygen supply and the removal of volatile compounds). The high C:N ratio enhances the immobilisation of NH<sub>4</sub>-N and thereby reduces NH<sub>3</sub> volatilisation (Dewes 1996, Kirchmann and Witter 1989). The lower level of available NH<sub>3</sub> restricts nitrification, while the aeration further hinders denitrification, reducing N<sub>2</sub>O production by 42-99%; additionally the higher

oxygen level impedes methanogenic activity, abating 45-99% of CH<sub>4</sub> emissions (Sommer and Moller 2000, Yamulki 2006). According to Fukumoto *et al.* (2003), stocking the manure into smaller pile sizes also increases the oxygenation rate of the heap, and leads to a decrease in both GHG and NH<sub>3</sub> emissions (67%, 77% and 64% of N<sub>2</sub>O, CH<sub>4</sub> and NH<sub>3</sub> emissions were eliminated, respectively).

Forced aeration also reduces the number and volume of anaerobic sites in the centre of the heap (Fukumoto *et al.* 2003), and controls GHG emissions (reductions up to 90%), although the NH<sub>3</sub> emission levels increase linearly with the air flow rate (Osada *et al.* 2000).

Similarly, the effect on NH<sub>3</sub> emissions of turning the manure heaps more frequently is not favourable, with 44-100% increase in the emissions (El Kader *et al.* 2007, Parkinson *et al.* 2004, Szanto *et al.* 2007) and substantial increase in the leached NH<sub>4</sub>-N as well (Parkinson *et al.* 2004), though Hassouna *et al.* (2008) found no significant effect on NH<sub>3</sub> emissions. At the same time, the results regarding nitrous oxide are inconclusive (Chadwick *et al.* 2011). CH<sub>4</sub> emissions are reported to be hugely decreased by frequent turning by one author (Szanto *et al.* 2007), and another research found no significant change in them (Hassouna *et al.* 2008).

## 2.7 Soil fertilisation

With the widespread use of inorganic fertilisers agricultural soils become an important source of N<sub>2</sub>O emissions, which is produced during the nitrification and denitrification of the N<sub>r</sub> added to the soils. Additionally, NH<sub>4</sub><sup>+</sup>-based synthetic fertilisers and urea, along with livestock manures spread on land and excreta deposited on pastures are significant sources of ammonia emissions. On the other hand, as most agricultural soils in Europe have predominantly oxidised environment, the CH<sub>4</sub> emissions observable after fertilisation originate from the CH<sub>4</sub> generated during the storage of liquid organic fertilisers (Sommer *et al.* 2009).

Synthetic fertiliser and manure spreading techniques which minimise the contact of manure with air are efficient ways to abate NH<sub>3</sub> emissions. Trailing hose spreaders apply slurry in narrow bands on top of the surface, while trailing shoe applicators have shoe-like attachments to deposit the slurry below the crop canopy. Injection techniques make shallow or deep cuts in the soil where slurry or other liquid fertiliser is placed. Finally, fertiliser can be incorporated into the soils by ploughing. These low-trajectory spreading techniques

decrease  $\text{NH}_3$ -N-losses, leaving more N available for subsequent processes, including nitrification and denitrification, and often producing wetter environment in the soils right around the fertiliser – increasing the likelihood of enhanced  $\text{N}_2\text{O}$  emissions. Nevertheless, this is not always the case – as summarised in reviews by Webb *et al.* (2010) and Chadwick *et al.* (2011). Webb *et al.* (2010) draw attention to the savings in indirect  $\text{N}_2\text{O}$  emission achieved by the  $\text{NH}_3$  abatement, which are likely to be higher than the increase in the direct  $\text{N}_2\text{O}$  emissions. As Chadwick *et al.* (2011) note, soil and climatic conditions favourable for denitrification (i.e. warm and/or wet weather, heavy soil structure and/or high moisture content) might result in increased  $\text{N}_2\text{O}$  emissions from slurry injection compared to broadcasting, but other conditions offer the opportunity of reducing  $\text{N}_2\text{O}$  and  $\text{NH}_3$  emissions simultaneously. Thorman *et al.* (2008) have a contrasting opinion, suggesting that conditions beneficial for denitrification might provide win-win situation for solid manure incorporation, while in other conditions incorporation is likely to increase  $\text{N}_2\text{O}$  emissions.

Using urease inhibitors along with urea fertiliser is another efficient way of reducing ammonia emissions from soils (Zaman *et al.* 2009). Though a meta-analysis of studies published in 2008 found no significant effect of urease inhibitors on  $\text{N}_2\text{O}$  emissions (Akiyama *et al.* 2010), in recent years many studies were published about their beneficial effects on  $\text{N}_2\text{O}$  emissions (Dawar *et al.* 2011, Halvorson *et al.* 2010, Halvorson *et al.* 2011, Sanz-Cobena *et al.* 2012, Vistoso *et al.* 2012).

Finally, changing the type of inorganic fertilisers can bring benefits for ammonia savings: urea has the highest potential for generating  $\text{NH}_3$  emissions, followed by  $\text{NH}_4^+$ -based fertilisers, while  $\text{NO}_3^-$ -based fertilisers generate the lowest  $\text{NH}_3$  emissions (Bussink and Oenema 1998, Misselbrook *et al.* 2000). However,  $\text{N}_2\text{O}$  emissions can be variable, as summarised by Snyder *et al.* (2009) and Harrison and Webb (2001), and Stehfest and Bouwman (2006) found no pronounced differences between most fertiliser types in terms of  $\text{N}_2\text{O}$  emissions after the sample had been balanced for other factors, like rate of application, crop type, climate, soil pH.

## 2.8 Conclusions

Inter-dependencies between agricultural  $\text{NH}_3$  and GHG emissions mean that almost all MMs have effects on more than one gaseous emission. Identifying the synergies and the trade-offs is crucial in supporting an integrated policy approach.

Higher production efficiency in livestock and crop production (using less input per unit of production) can reduce most of the environmental burdens arising from agricultural production, including  $\text{NH}_3$  and GHG pollution per unit of crop and/or livestock output. Examples include improved cattle fertility, stricter pest and disease monitoring, and optimised grazing management. Ongoing genetic selection in crops and animals can advance resource use efficiency in a variety of goals, including such as N, energy and water use efficiency, disease resistance, fertility and longevity. To best adapt new varieties and breeds to their environment, climate change is an important consideration when employing genetic selection.

Animal feeding techniques targeting N input are usually win-win solutions for  $\text{NH}_3$  and  $\text{N}_2\text{O}$ , and in some cases also for  $\text{CH}_4$ , though some feeding techniques, e.g. higher starch or sugar content, can provoke land use change, with negative implications on biodiversity, food security and lifecycle GHG emissions from land converted to croplands.

Livestock system changes (changing the proportion of the time spent outdoors or changing between solid and liquid manure systems) can result in pollution swapping: the reduced  $\text{NH}_3$  and  $\text{CH}_4$  emissions are accompanied by increased  $\text{N}_2\text{O}$  emissions or vice versa. The majority of  $\text{NH}_3$  emissions from agriculture originate from bedding and manure storage, while  $\text{N}_2\text{O}$  emissions from the same source contribute only 6% to total agricultural GHG in Europe (Table 3), the evident trade-offs should be considered. Furthermore for both solid and liquid manure systems various efficient ammonia and GHG MMs are available, some providing savings in both pollutants. But when comparing the different housing MMs it is important to consider that most of the feeding, housing, and manure storage MMs are unavailable for the time what animals spend outdoors.

Changed housing design and in-house manure management practices can offer both  $\text{NH}_3$  and GHG benefits. Win-win MMs for both liquid and solid manure storage exist, including slurry acidification and airtight covering of solid manure heaps. Nevertheless, many other MMs targeting manure storage show variable results for the different gases, making the outcomes uncertain.

To reduce  $\text{NH}_3$  emissions from soil fertilisation, low trajectory manure spreading and urease inhibitors are important MMs. The former has uncertain effects on  $\text{N}_2\text{O}$  emissions, the effects depending on local conditions. The latter might prove to be an option to reduce both GHG and  $\text{NH}_3$  emissions.

The most promising win-win measures are improving production efficiency and N-use efficiency. Low-emission housing design and management (with attention to all types of gaseous emissions) is also likely to deliver multiple benefits and is becoming more important due to the increasing concentration of livestock production and an emerging need for housing more adaptable to the changing climate. Slurry acidification, urease inhibitors and the choice of inorganic fertilisers are also potential win-win MMs.

There is a risk of pollution swapping when the amounts of starch and sugar in animal feeds are increased, when changing indoor/outdoor housing and liquid/solid manure management systems, from separating slurry and from increasing the aeration of solid manure. When choosing between these alternatives and current practices, the negative and positive effects of the different pollutants have to be weighted and compared. An example assessment considering multiple pollutants is presented in Section 3.

Some MMs require further investigation; for example low-trajectory manure spreading could be a win-win solution in some circumstances. The effectiveness of covering slurry stores and manure heaps is highly dependent on the type of material and method used, and could offer opportunities for a concurrent reduction in GHGs and  $\text{NH}_3$ . Anaerobic digestion of animal waste has important positive consequences beyond the farm gate, and might be a win-win measure if efficient ammonia MMs are applied in the storage and spreading of the digestate and if the substrates do not contribute to reducing carbon stocks via land use change.

In many cases the MMs have effects on the whole farm, potentially impacting on yield, product quality or gaseous emissions from other parts of the system. Whole-farm biophysical and economic models can help understanding these interdependencies. Beyond the farm gate changes are also possible. For example reduced grass fertilisation rates imply lower synthetic fertiliser production and therefore reduced  $\text{CO}_2$  and  $\text{N}_2\text{O}$  emissions from industrial processes. Optimising the diet can also lead to off-farm emission changes from fertiliser related emissions of feed crops or in the soil carbon stock if the land use pattern changes. Beyond gaseous emissions, the financial costs, other – usually locally and regionally important – environmental effects (e.g. biodiversity, water pollution, soil degradation) and social consequences have to be considered.

### 3 Paper II. Multiple-pollutant cost-effectiveness of greenhouse gas mitigation measures in the UK agriculture

Vera Eory, Cairistiona F. E. Topp, Dominic Moran

Research Division, SRUC, West Mains Road, Edinburgh EH9 3JG, UK

This piece of research was published in *Environmental Science & Policy* in 2013, vol 27, pp. 55-67.

My contribution to the research was the design and implementation of the methodology and the analysis and discussion of results.

#### 3.1 Abstract

This paper develops multiple-pollutant marginal abatement cost curve analysis to identify an optimal set of greenhouse gas mitigation measures considering the trade-offs and synergies with other environmental pollutants. The analysis is applied to UK agriculture, a sector expected to make a contribution to the national greenhouse gas mitigation effort. Previous analyses using marginal abatement cost curves have determined the sector's greenhouse gas abatement potential based on the cost-effectiveness of a variety of technically feasible mitigation measures. Most of these measures have external effects on other pollution loads arising from agricultural activities. Here the monetary values of four of the most important impacts to water and air (specifically ammonia, nitrate, phosphorous and sediment) are included in the cost-effectiveness analysis. The resulting multiple-pollutant marginal abatement cost curve informs the design of sustainable climate change policies by showing how the multiple-pollutant marginal abatement cost curve for the UK agriculture can differ from the greenhouse gas marginal abatement cost curve. The analysis also highlights research gaps, and suggests a need to understand the wider environmental effects of greenhouse gas mitigation measures and to reduce the uncertainty in pollutant damage cost estimates.

## 3.2 Introduction

Greenhouse gas mitigation is on the top of the environmental policy agenda as countries seek to meet emissions reduction commitments. Agriculture is an important source of GHG emissions, accounting for 10-12% of total global and 9% of UK GHG emissions (Smith *et al.* 2007, Thomas *et al.* 2011). The sector is thought to offer significant emission reduction potential through the deployment of a number of cost-effective mitigation and carbon sequestration measures. But the implementation of these measures can occasion other environmental impacts that need to be addressed in any overall assessment of measure CE.

Land based MMs can be highly variable in terms of their emission reduction (abatement) potential and private cost of measure implementation. Moreover, some measures have wider environmental co-effects (external effects), that can be both positive and negative. Adding these co-effects to the private cost of measures defines a social cost that can be used to redefine the CE of measures (i.e. the costs of implementation relative to GHG benefits). This paper investigates the social cost of GHG MMs and aims to outline a more accurate CE metric for ranking measures in a MACC.

MACCs are tools to identify relatively cost-effective MMs across the economy (Kesicki and Strachan 2011). MACCs can also be used to define the economically optimal level of abatement, where marginal abatement costs are equal to the resulting marginal benefits (Pearce and Turner 1989). In practice, the economically optimal level of GHG abatement is defined by comparing marginal abatement costs with a standard marginal benefit benchmark such as the shadow price of carbon (SPC).

Figure 5 shows how adding external effects can alter the MACC. Positive co-effects reduce abatement costs, while negative ones increase the abatement cost, thus tilting the curve. The intercept of the MACC and the marginal benefit curve indicates the economically optimal level of abatement ( $q^*$ ). In case where the co-effects are mostly positive, the reduced abatement costs result in an increased abatement optimum ( $q'$ ), or in decreased overall costs of achieving a targeted pollution reduction level.



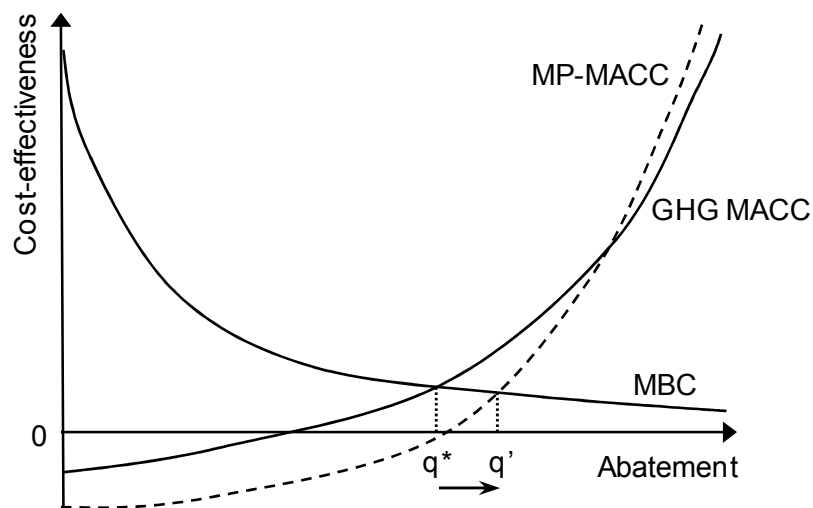


Figure 5. Effect of inclusion of co-effects on the GHG MAC curve and on the economic optimum of pollution control

$q^*$ ,  $q'$ : economic optimum of pollution reduction without and with co-effects, respectively; MBC: marginal benefit curve; MP-MACC: multiple-pollutant MACC

GHG MACCs have been constructed for various sectors including energy and transport (Enkvist *et al.* 2007), and have galvanised wider debate and action on mitigation policy. MACCs have also been used to inform policy development on measures targeting various agricultural pollutants (see e.g. Webb *et al.* (2006) for ammonia, Haygarth *et al.* (2009) for phosphorous and Scholefield *et al.* (2004) for nitrates). But these studies have been limited in their treatment of any co-effects and hence of trade-offs and synergies between different agricultural pollutants (Reis *et al.* 2005).

There is a growing literature on modelling multiple pollutants. Brink *et al.* (2001, 2005) analysed the co-effects of  $\text{NH}_3$  and GHG MMs in European agriculture. Wagner *et al.* (2012) presented a multi-sector assessment of GHG MMs and their air pollution co-effects ( $\text{SO}_2$ , mono-nitrogen oxides, fine particulate matter ( $\text{PM}_{2.5}$ )) in Annex I countries to the United Nations Framework Convention on Climate Change. Anthony *et al.* (2008) provided a cost-benefit assessment of six agricultural pollutants ( $\text{NO}_3^-$ , P, sediment,  $\text{NH}_3$ ,  $\text{CH}_4$  and  $\text{N}_2\text{O}$ ) for the UK. In the US, Schneider *et al.* (2007) estimated the external effects of GHG MMs on soil erosion, N and phosphorous (P) pollution. The optimisation approach in these studies is either based on a single pollutant, or provides the least-cost solution based on specified pollution reduction targets.

In contrast, a MACC can potentially facilitate the representation of the socially optimal abatement potential by accommodating multiple pollutants into a marginal cost curve. This single metric can be generated by monetising environmental co-effects, creating a multiple-pollutant MACC. Relative to a GHG MACC, a multiple-pollutant marginal abatement cost curve (MP-MACC) also enables better representation of the social cost of integrated policies.

This paper considers the consequences of including available data on the monetary valuation of GHG MMs' co-effects into the existing GHG MACC estimates developed for agriculture in the UK (Moran *et al.* 2011b). The external effects included are NO<sub>3</sub> leaching, NH<sub>3</sub> emissions, P and sediment pollution. We are unaware of any studies adding co-effects of mitigation effort to MACCs using a single metric of CE.

The rest of the paper is structured as follows. Section 3.3 provides more background to the MP-MACC analysis in agriculture. Sections 3.4 and 3.5 outline a methodology for the paper and present results. Sections 3.6 and 3.7 provide a discussion and a conclusion, respectively.

### 3.3 Background

Agriculture is expected to make a contribution to the national GHG mitigation effort in the UK that is being coordinated by the UK Committee on Climate Change and partly informed by sector-wide MACC analyses. Technically feasible measures for mitigating GHG emissions in the UK agriculture include, for example, improved resource use efficiency at farm level, generating greater output per unit of input. Higher efficiency can be achieved via selective breeding of livestock, optimised feeding strategies and judicious use of nitrogen fertilisers. Other MMs include changes in animal housing and manure storage, enhancing the removal of atmospheric CO<sub>2</sub> via sequestration into soil and vegetation sinks and replacing fossil fuel emissions with alternative energy sources.

Earlier GHG MACC analysis identified a financially feasible subset of measures, based on the private costs of implementation and on the abatement potential of the measures (Moran *et al.* 2011a). The analysis noted the particular biophysical complexities of agricultural mitigation and the likelihood of potentially large co-effects associated with the widespread implementation of many measures. These co-effects could include reduced (or increased) pollution to water, mitigation of other pollutants including NH<sub>3</sub>, and more complex impacts to ecosystems functions.

Specific effects considered in this analysis are  $\text{NO}_3^-$  leaching,  $\text{NH}_3$  emissions, P and sediment pollution. These pollutants are drivers of environmental changes, leading to changes in ecosystem services.  $\text{NO}_3^-$  and P cause eutrophication in aquatic ecosystems, and drinking water  $\text{NO}_3^-$  levels are controlled to eliminate the risk of methaemoglobinaemia. Sediment in water-bodies (originating from soil erosion) has negative effects on biological water quality, contributes to drinking water contamination, and when deposited by fluvial flooding, can damage property, roads and transport links. Water-borne sediment is only part of the problem arising from soil erosion; other effects of soil erosion are not included in this assessment.  $\text{NH}_3$  emissions are associated with human health and environmental issues, most importantly respiratory problems (via the formation of secondary aerosols contributing to particulate matter concentrations above critical levels), acidification and eutrophication (both aquatic and terrestrial). For reviews on processes related to  $\text{NO}_3^-$  and  $\text{NH}_3$  see, for example Chapter 22 and 23 of The European Nitrogen Assessment (Brink *et al.* 2011, Oenema *et al.* 2011), for P Correll (1998), and for sediment, Pimentel *et al.* (1995).

Accordingly, the current study draws on existing evidence on both the biophysical impact of other pollutants and available damage costs. Damage costs are monetary estimates of the damage a pollutant causes to society. Here these effects are quantified in monetary terms using evidence from existing non-market valuation literature that can be transferred for use in the MP-MACC.

## 3.4 Methods

### 3.4.1 Calculations

In this paper we further develop the GHG MACC elaborated in Moran *et al.* (2008), where the MACC analysis ranks the MMs in decreasing order of CE by dividing the cost of the measure with the GHG abatement achievable. The measures are additional to mitigation activity that would be expected to happen in a BAU baseline. That analysis was revisited to represent uncertainty of effectiveness assumptions and to further develop existing interaction factors between the measures to avoid double-counting the abatement potential of individual measures (MacLeod *et al.* 2010b). In that paper alternative empirical estimates were used to approximate uncertainties, leading to the construction of optimistic and pessimistic abatement scenarios based on upper and lower abatement rates, applicability rates and cost

estimates for specific MMs. A maximum technical potential refers to the level of abatement reached assuming full uptake by farmers. In contrast, a lower feasible potential allows for the likelihood of behavioural constraints that suggest that no MMs are likely to be taken up to 100%. We use the 2022 maximum technical potential (MTP) Optimistic MACC as a basis for the current analysis.

The CE calculation underlying the GHG MACC is initially altered to accommodate the additional net external effects associated with each MM. For the MP MACC the quantitative emission reduction of the MM on each pollutant are multiplied with the damage costs of the pollutants to derive an estimate of the monetary value of the external effects (Equation 1). The external effects are then added to the private cost of the MM, providing the social cost of the measure, which is used in the social CE calculation (Equation 2). From this point on, the calculation follows the method described in (Moran *et al.* 2008) and (MacLeod *et al.* 2010b).

*Equation 1*

$$External\ cost_i = \sum_{j=1}^k (Change\ in\ pollution\ load)_{i,j} * (Damage\ cost)_j$$

Where:

*External cost<sub>i</sub>*: monetary value of the annual external effects of MM *i* (£ y<sup>-1</sup>)

*Change in pollution load<sub>i,j</sub>*: change in the annual pollution of pollutant *j* caused by MM *i* (t y<sup>-1</sup> NO<sub>3</sub><sup>-</sup>-N, NH<sub>3</sub>-N, P and sediment, respectively)

*Damage cost<sub>j</sub>*: damage cost of pollutant *j* (£ t<sup>-1</sup> NO<sub>3</sub><sup>-</sup>-N, NH<sub>3</sub>-N, P and sediment, respectively)

*Equation 2*

$$Social\ CE_i = \frac{(Private\ cost)_i + (External\ cost)_i}{(GHG\ saved)_i}$$

Where:

*Social CE<sub>i</sub>*: CE of MM *i* with external effects

*Private cost<sub>i</sub>*: private cost of implementation of MM *i*

*GHG saved<sub>i</sub>*: GHG mitigation by MM *i*

### 3.4.2 Data sources

MacLeod *et al.* (2010b) provided data on the range of MMs, their GHG abatement rates and private costs along with interaction factors. In that paper, baseline activity data (animal numbers, land areas) represent 2022 forecasts derived from Shepherd *et al.* (2007).

Applicability and GHG abatement rates of the measures are based on a literature review and expert opinion (MacLeod *et al.* 2010b). Cost data for crop and soil measures are derived from a representative farm optimisation model (see more in (MacLeod *et al.* 2010a)), while livestock costs are adopted from IGER (2001).

The quantity of associated co-effects of measures on  $\text{NO}_3^-$ ,  $\text{NH}_3$  P and sediment pollution were reported in Anthony *et al.* (2008), derived from the application of a range of process models: NARSES (Webb and Misselbrook 2004), NT26AE (Chadwick *et al.* 2005), MANNER (Chambers *et al.* 1999), PSYCHIC (Davison *et al.* 2008), NEAP-N (Lord and Anthony 2000), NITCAT (Lord 1992). In that paper the estimated pollution loads were based on activity data for 2004 (Shepherd *et al.* 2007), the pollution saving values reported account for interactions between the measures to avoid double counting of the pollution reduction. The different base years of the two datasets (Anthony *et al.* 2008, MacLeod *et al.* 2010b) lead to a slight discrepancy in the current analysis, resulting in approximately 5% underestimation of external effects for measures applicable to arable land and similar overestimation of co-effects for livestock measures. Note also that the Anthony *et al.* (2008) study only covered pollution loading in England and Wales and we therefore restrict the subsequent analysis to England and Wales.

The attribution of external effects to MMs required a comparison of the specific MMs being evaluated in the two key studies (Anthony *et al.* 2008, MacLeod *et al.* 2010b). A match between the measures was achieved by either amalgamating or disaggregating some measures in one or other of the studies. In some cases no match was possible, resulting in data gaps for some of the measures on the MP MACC (Table 5).

Table 5. Matching measures between the two studies

(MacLeod <i>et al.</i> 2010b)		(Anthony <i>et al.</i> 2008)	
ID <sup>1</sup>	Measure name	ID	Measure name
AA	Using biological fixation to provide N inputs (clover)	31	Use clover in place of grass
AD	Avoiding N excess	22	Use a fertiliser recommendation system
AE	Full allowance of manure N supply	23	Integrate fertiliser and manure nutrient supply

(MacLeod <i>et al.</i> 2010b)		(Anthony <i>et al.</i> 2008)	
ID <sup>1</sup>	Measure name	ID	Measure name
AG	Improved timing of mineral fertiliser N application	26	Avoid spreading fertiliser to fields at high-risk times
AJ	Improved timing of slurry and poultry manure application	71	Do not spread slurry at high-risk times
		74	Do not spread solid manure to fields at high-risk times
		<i>Results added up.</i>	

<sup>1</sup> ID refers to the measures' codes in these studies (Anthony *et al.* 2008, MacLeod *et al.* 2010b)

Table 5. cont.

(MacLeod <i>et al.</i> 2010b)		(Anthony <i>et al.</i> 2008)	
ID <sup>1</sup>	Measure name	ID	Measure name
AL	Plant varieties with improved N-use efficiency	20	Use plants with improved nitrogen use efficiency
AN	Reduced tillage / No-till	7	Adopt reduced cultivation systems
BA	Increased high starch concentrate in diet, dairy	33	Reduce dietary nitrogen and phosphorus intakes
BB	Increased maize silage in diet, dairy	<i>Notes:</i> 1. The P-effect of Measure 33 was ignored. 2. Most pig and poultry farms already use high quality protein feeds, therefore Measure 33 mainly refer to changes in cattle feeding. 3. In cattle the main methods of reducing the N content of the diet are increasing the starch and reducing the grass/forage content of the feed by increasing the proportion of starchy concentrates or maize silage.	
CA	Increased high starch concentrate in diet, beef		
<i>BA, BB and CA are merged into two new measures to cover all cattle (note that BA and BB are mutually exclusive):</i>			
BACA	Increased high starch concentrate in diet, dairy+beef		
BBCA	Increased maize silage for dairy and increased high starch concentrate for beef		
BF	Improved genetic potential for dairy cows – productivity		
BI	Improved genetic potential for dairy cows – fertility		
CG	Improved genetic potential for beef cattle		
<i>Merged into a new measure:</i>			
KA	Improved cattle genetics	56	Install covers to slurry stores
FA	Covering lagoons – dairy		
FB	Covering slurry tanks – dairy		
GA	Covering lagoons – beef		
GB	Covering slurry tanks – beef		
IA	Covering lagoons – pigs		
IB	Covering slurry tanks – pigs	58	Anaerobic digestion
EB-EI	On-farm anaerobic digestion measures (dairy/beef/pig, medium/large farms)		
HA-HT	Centralised anaerobic digestion measures (dairy/beef/pigs/poultry, 1MW/2MW/3MW/4MW/5MW)		
AB	Reduce N fertiliser	-	<i>No matching measure.</i>
AC	Land drainage	-	<i>No matching measure.</i>
AF	Species introduction (including legumes)	-	<i>No matching measure.</i>
AH	Controlled release fertilisers	-	<i>No matching measure.</i>
AI	Nitrification inhibitors	-	<i>No matching measure.</i>
AK	Adopting systems less reliant on inputs	-	<i>No matching measure.</i>

<sup>1</sup> ID refers to the measures' codes in these studies (Anthony *et al.* 2008, MacLeod *et al.* 2010b)

Table 5. cont.

(MacLeod <i>et al.</i> 2010b)		(Anthony <i>et al.</i> 2008)	
ID <sup>1</sup>	Measure name	ID	Measure name
AM	Separate slurry applications from fertiliser applications by several days	-	No matching measure.
AO	Use composts, straw-based manures in preference to slurry	-	No matching measure.
BE	Ionophores, dairy	-	No matching measure.
BG	Bovine somatotropin, dairy	-	No matching measure.
BC	Propionate precursors, dairy	-	No matching measure.
BD	Probiotics, dairy	-	No matching measure.
BH	Transgenic manipulation of ruminants, dairy	-	No matching measure.
CC	Propionate precursors, beef	-	No matching measure.
CD	Probiotics, beef	-	No matching measure.
CE	Ionophores, beef	-	No matching measure.
FC	Switch from anaerobic to aerobic storage - dairy slurry tanks	-	No matching measure.
FD	Switch from anaerobic to aerobic storage - dairy lagoons	-	No matching measure.
GC	Switch from anaerobic to aerobic storage - beef slurry tanks	-	No matching measure.
GD	Switch from anaerobic to aerobic storage - beef lagoons	-	No matching measure.
IC	Switch from anaerobic to aerobic storage - pigs slurry tanks	-	No matching measure.
ID	Switch from anaerobic to aerobic storage - pigs lagoons	-	No matching measure.

<sup>1</sup> ID refers to the measures' codes in these studies (Anthony *et al.* 2008, MacLeod *et al.* 2010b)

Monetary estimates are required to value the emission reduction of each pollutant associated with the measures. To reflect the uncertainty in the damage costs the current analysis uses five sets of damage costs derived from the literature and recent policy reports, covering the four pollutants: NO<sub>3</sub><sup>-</sup>, P, sediment, NH<sub>3</sub> (Table 6). These estimates were applied to the 2022 MTP Optimistic GHG MACC. Sets A and B are described by Anthony *et al.* (2008), while sets C, D and E were added to the analysis to represent the higher end of damage cost estimates found in the literature. The 2022 MTP Optimistic GHG MACC is considered in the analysis for comparison.



Table 6. Unit damage cost sets used in this study

Damage value set	NO <sub>3</sub> -N [£ t <sup>-1</sup> ]	P [£ t <sup>-1</sup> ]	Sediment [£ t <sup>-1</sup> ]	NH <sub>3</sub> -N [£ t <sup>-1</sup> ]	MACC
None	-	-	-	-	GHG-MACC
A	217 <sup>1</sup>	9,634 <sup>1</sup>	25 <sup>1</sup>	1,804 <sup>2</sup>	MP-MACC-A
B	672 <sup>3</sup>	45,144 <sup>3</sup>	108 <sup>3</sup>	1,804 <sup>2</sup>	MP-MACC-B
C	4,287 <sup>4</sup>	9,634 <sup>1</sup>	25 <sup>1</sup>	17,699 <sup>5</sup>	MP-MACC-C
D	4,287 <sup>4</sup>	45,144 <sup>3</sup>	108 <sup>3</sup>	17,699 <sup>5</sup>	MP-MACC-D
E	20,577 <sup>4</sup>	45,144 <sup>3</sup>	108 <sup>3</sup>	52,055 <sup>5</sup>	MP-MACC-E

<sup>1</sup> values derived by (Anthony *et al.* 2008) after Spencer *et al.* (Spencer *et al.* 2008)

<sup>2</sup> Defra damage cost (Defra 2008) as used in (Anthony *et al.* 2008)

<sup>3</sup> values derived by (Anthony *et al.* 2008), based on (Baker *et al.* 2007, Spencer *et al.* 2008)

<sup>4</sup> values based on (Brink *et al.* 2011)

<sup>5</sup> values based on (Holland *et al.* 2005)

The sources for the selected damage costs for each external effect are shown in Table 7, which also reflects a variety of methodological approaches to derive monetary values. We note that this form of non-market valuation is problematic both in terms of the differences in methodological approach and the paucity of studies for specific co-effects relating to the pressures under consideration. In general, this means that some form of benefits transfer is necessary (see (Brouwer 2000)), which inevitably introduces some subjectivity into how well existing studies match the external effects under investigation. The carbon price threshold used in this paper to represent the marginal benefit of mitigation is £34.3 t CO<sub>2</sub>e<sup>-1</sup> in 2022 (Price *et al.* 2007).

Table 7. Basis of valuation of damage costs

Pollutant	Damage cost set	Impact	Basis of valuation	References
NO <sub>3</sub> <sup>-</sup> , P, sediment	NO <sub>3</sub> <sup>-</sup> : A/B P, sediment: A/B/C/D/E	NO <sub>3</sub> <sup>-</sup> in drinking water	Expenditure on nitrate removal by water companies.	(Spencer <i>et al.</i> 2008)
		Particulates in drinking water	Expenditure on sediment removal by water companies.	(Spencer <i>et al.</i> 2008)
		Agricultural pollution incidents	Costs of restocking rivers with fish following a pollution incident.	(Spencer <i>et al.</i> 2008)
		Degraded river quality	Benefit transfer of willingness to pay for water quality of the River Tame.	(Georgiou <i>et al.</i> 2000, Spencer <i>et al.</i> 2008)
		Degraded estuary quality	Benefit transfer of willingness to pay for water quality of the River Tame.	(Georgiou <i>et al.</i> 2000, Spencer <i>et al.</i> 2008)
		Lake eutrophication	Reduced value of waterside property, reduced value of water for commercial use, drinking water treatment costs (algae), clean-up costs of waterways, increased GHG and acidifying gases emissions, reduced recreational amenity value, overall ecological damage.	(O'Neill 2007, Pretty <i>et al.</i> 2002, Spencer <i>et al.</i> 2008)
		Offsite soil erosion damage	Costs of dredging waterways, damage to property and roads.	(Spencer <i>et al.</i> 2008)
NO <sub>3</sub> <sup>-</sup>	C/D/E	Degraded marine quality	Willingness to pay for clean Baltic Sea. Higher and lower bound.	(Brink <i>et al.</i> 2011, Gren <i>et al.</i> 2008, Soderqvist and Hasselstrom 2008)
		Nitrates in drinking water	Mortality from colon cancer. Higher and lower bound.	(Brink <i>et al.</i> 2011, van Grinsven <i>et al.</i> 2010)
NH <sub>3</sub>	A/B	Secondary inorganic aerosols	Chronic mortality caused by PM <sub>2.5</sub> .	(Defra 2008)
NH <sub>3</sub>	C/D/E	Secondary inorganic aerosols	Chronic mortality, chronic and acute morbidity caused by PM <sub>2.5</sub> , along with effects on crops by hindering tropospheric ozone formation.	(Holland <i>et al.</i> 2005)

## 3.5 Results

Results are first presented on the value of co-effects for each measure where data are available. The analysis will then demonstrate how these benefits affect the shape of the GHG MACCs and hence the overall cost-effective abatement potential.

### 3.5.1 Private and external cost comparison

Annual private cost of the measures falls in a range of £-811 million, (“Ionophores, beef”) to £1,650 million (“Transgenic manipulation of ruminants, dairy”), negative values denoting a saving. Recall that this refers to the extent of cost/savings resulting from the application of a MM to its full extent, i.e. across the whole English and Welsh agriculture. The annual value of external effects varies from £-16 million (“Improved cattle genetics”) to £0 (anaerobic digestion measures) calculated with damage cost set A, and from £-512 million (“Plant varieties with improved N-use efficiency”) to £0 (anaerobic digestion measures) calculated with damage cost set E (Figure 6). The external effects are net positive for measures where data are available, because none have any negative co-effects on the pollutants considered. The external effects have generally higher values for crop and soil measures and measures on cattle feeding and genetics, and are lower for manure management measures (covering slurry stores) and zero for anaerobic digestion measures. Changing between damage cost sets from A to E increases the amount of external benefits. The two biggest increases in the external benefits arise when changing from damage cost set B to C and damage cost set D to E, where  $\text{NH}_3$  and  $\text{NO}_3^-$  damage costs are increased considerably. Changing damage costs from set A to B or C to D (where the damage costs of P and sediment are increased) does not have a big impact on the value of co-effects.

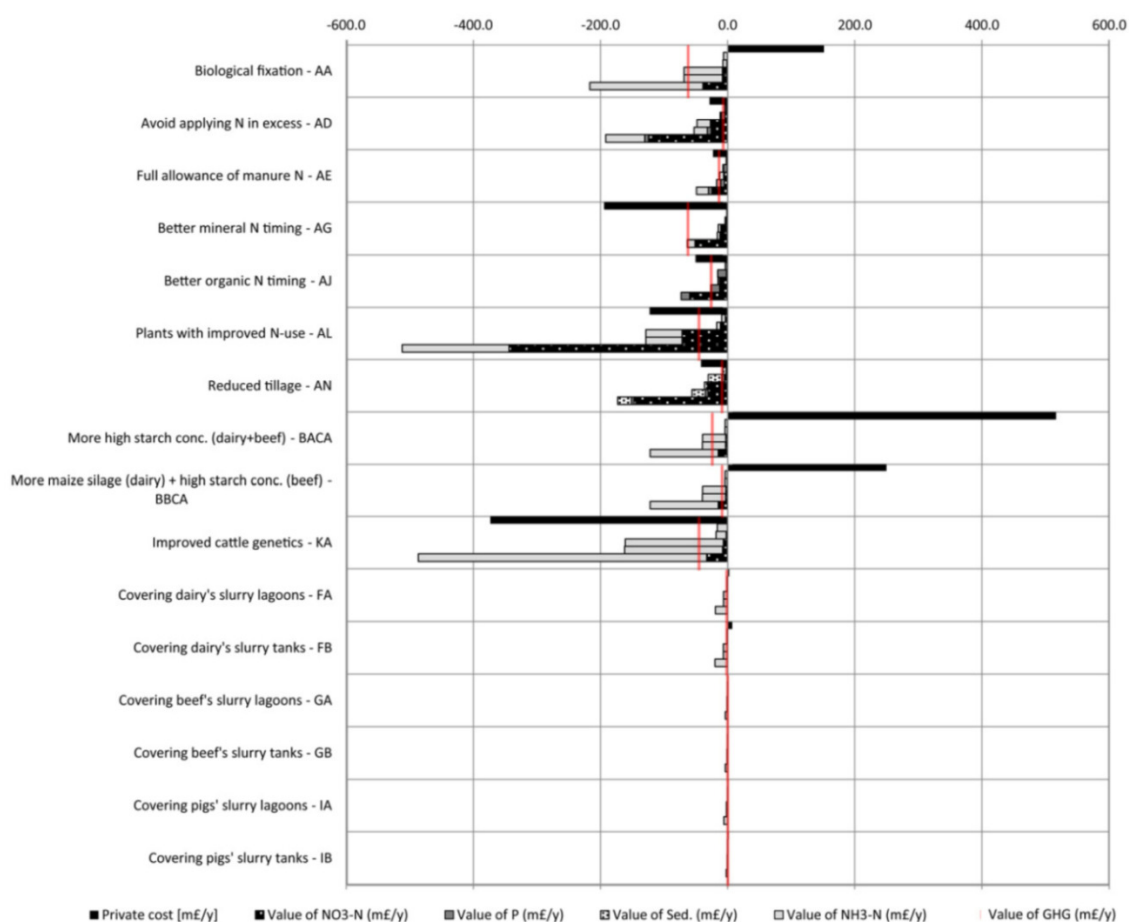


Figure 6. Annual private and social costs and the value of GHG savings.

Measures are shown where data are available on external effects (with the exception of anaerobic digestion measures, as the value of their co-effects are zero). The contribution of each co-effect to the social costs is indicated by separate patterns. For each measure the six bars from top to bottom represent the private costs, social costs from damage cost set of A, B, C, D and E, respectively. The monetary value (calculated by using the SPC) of the GHG savings are shown by red vertical lines. Sed.: sediment

### 3.5.2 GHG-MACC

With no co-effects, the economically optimal GHG abatement (i.e. measures with CE below the shadow price of carbon), assuming full uptake of measures by the farmers, is 11.9 Mt CO<sub>2</sub>e y<sup>-1</sup> for England and Wales in 2022. The total abatement potential of measures with negative costs (i.e. measures' CE ≤ £0) is 11.8 Mt CO<sub>2</sub>e y<sup>-1</sup>. This is 36% of agricultural GHG emissions in England and Wales, which are expected to be 32.6 Mt CO<sub>2</sub>e in that year (Defra 2011).

### 3.5.3 MP-MACC-A and MP-MACC-B

Adding the effect of NO<sub>3</sub><sup>-</sup>, NH<sub>3</sub>, P and sediment valued with the damage cost sets A and B to the private costs of the measures has a small effect on the MACC (Figure 7). The CE improves slightly for all measures that have data on co-effects (except anaerobic digestion measures). The cumulative GHG abatement of measures with CE ≤ 0 increases only narrowly, by 14 kt CO<sub>2</sub>e y<sup>-1</sup>. Counter intuitively, the cumulative GHG abatement of measures with CE ≤ SPC is reduced by 110 kt CO<sub>2</sub>e y<sup>-1</sup>. This is due to on-farm pig anaerobic digestion measures being replaced by covering of slurry tanks and lagoons (pigs), as the latter became more cost-effective. Covering pig slurry tanks and lagoons provides smaller abatement potential than on-farm anaerobic digestion of pig manure.

MP-MACC-B shows only small differences from MP-MACC-A in spite of a three to five-fold increase in the damage cost of NO<sub>3</sub><sup>-</sup>, P and sediment, because the social benefits are still far less than the absolute value of the private costs/benefits.

For both damage cost sets A and B, the annual abatement potential under CE = 0 for the non-GHG pollutants are 38 kt NO<sub>3</sub><sup>-</sup>-N, 0.7 kt P, 198 kt sediment and 14 kt NH<sub>3</sub>-N (14%, 18%, 11% and 9% of annual load from agriculture, respectively). Total annual loads are estimated by Anthony *et al.* (2008) to be 276 kt nitrate, 4.0 kt phosphorous, 1,790 kt sediment and 158 kt NH<sub>3</sub> in 2020. The pollutant reduction results are the same when the CE threshold is increased from 0 to the SPC.

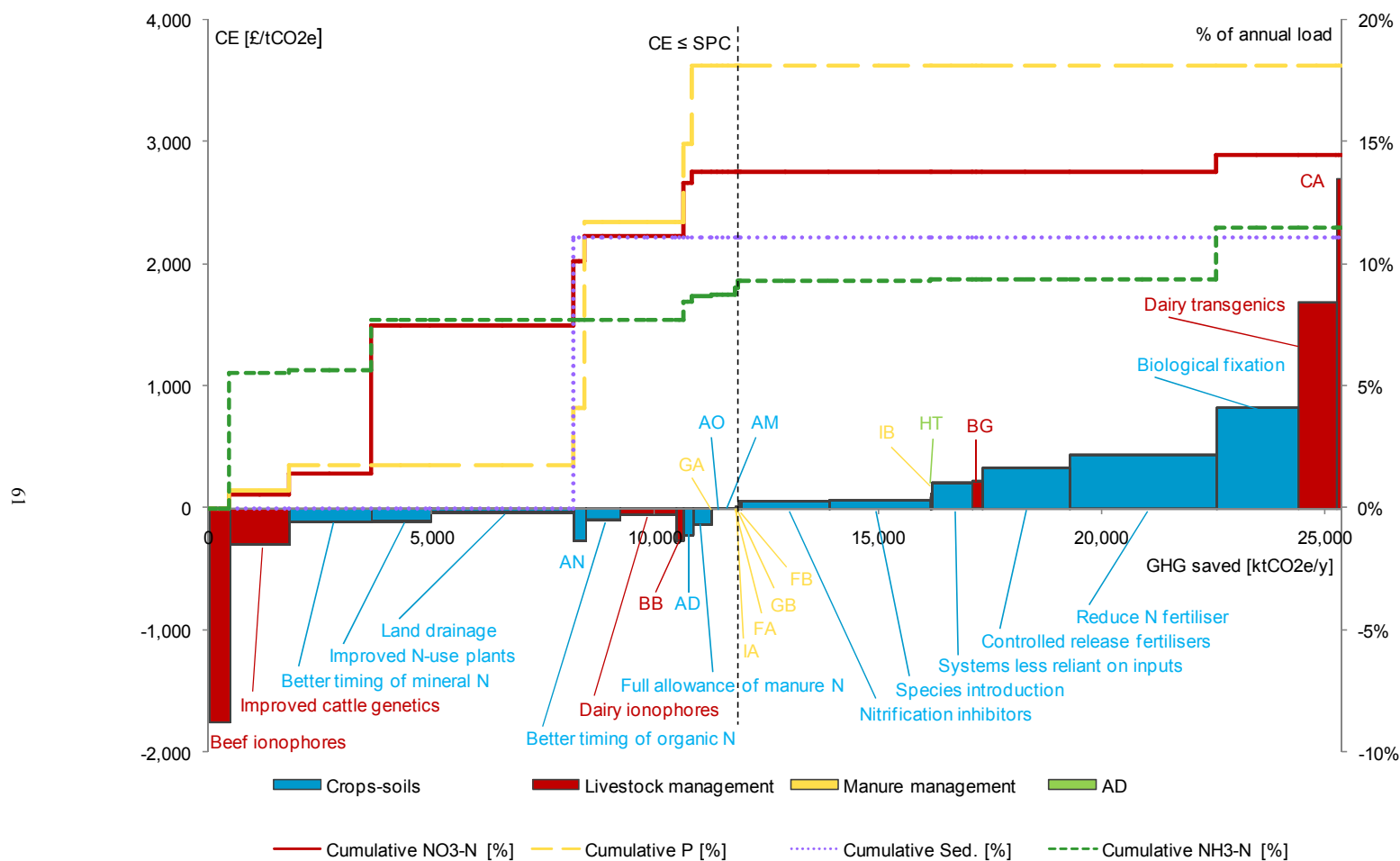


Figure 7. MP-MACC-B

The bars represent the MP-MACC while the lines represent the cumulative savings in the annual load of the four pollutants. Sed.: sediment. See Table 5 for the names of the measures with abatement potential less than 400 kt CO<sub>2</sub>e y<sup>-1</sup>

### 3.5.4 MP-MACC-C and MP-MACC-D

Applying higher damage costs, again leads to a slight change in the MACC, increasing the GHG abatement potential by 1% (MP-MACC-C or MP-MACC-D compared to MP-MACC-A or MP-MACC-B). The biggest difference from MP-MACC-A and MP-MACC-B is that the CEs of all but one of the slurry store covering MMs are now below the SPC. Both the GHG and NH<sub>3</sub> savings are improved by this change (by 0.07 Mt CO<sub>2</sub>e and 0.4 kt NH<sub>3</sub>-N, respectively). On these MACCs, only two measures for which we have data on external effects remain economically inefficient (having CE > SPC): “Using biological fixation to provide N inputs” and “Centralised anaerobic digestion, poultry, 5MW”.

Again, the large increase in the value of P and sediment, in MP-MACC-D relative to MP-MACC-C has only a slight effect, improving the CE of four measures that already had negative CEs.

### 3.5.5 MP-MACC-E

With higher damage costs for NO<sub>3</sub><sup>-</sup> and NH<sub>3</sub> “Using biological fixation to provide N inputs” becomes economically efficient, with the cumulative annual GHG savings increasing by 1.8 Mt to 13.8 Mt CO<sub>2</sub>e, and the NH<sub>3</sub> savings by 3.4 kt to 18.2 kt NH<sub>3</sub>-N for both a CE threshold of zero or SPC (Figure 8).

As shown in Figure 9, introducing damage costs for measures where co-effect data are available improves the CE of all measures except for the anaerobic digestion measures, which have no effect on NH<sub>3</sub>, NO<sub>3</sub><sup>-</sup>, P or sediment pollution. The data available on external effects imply that no pollution swapping occurs.

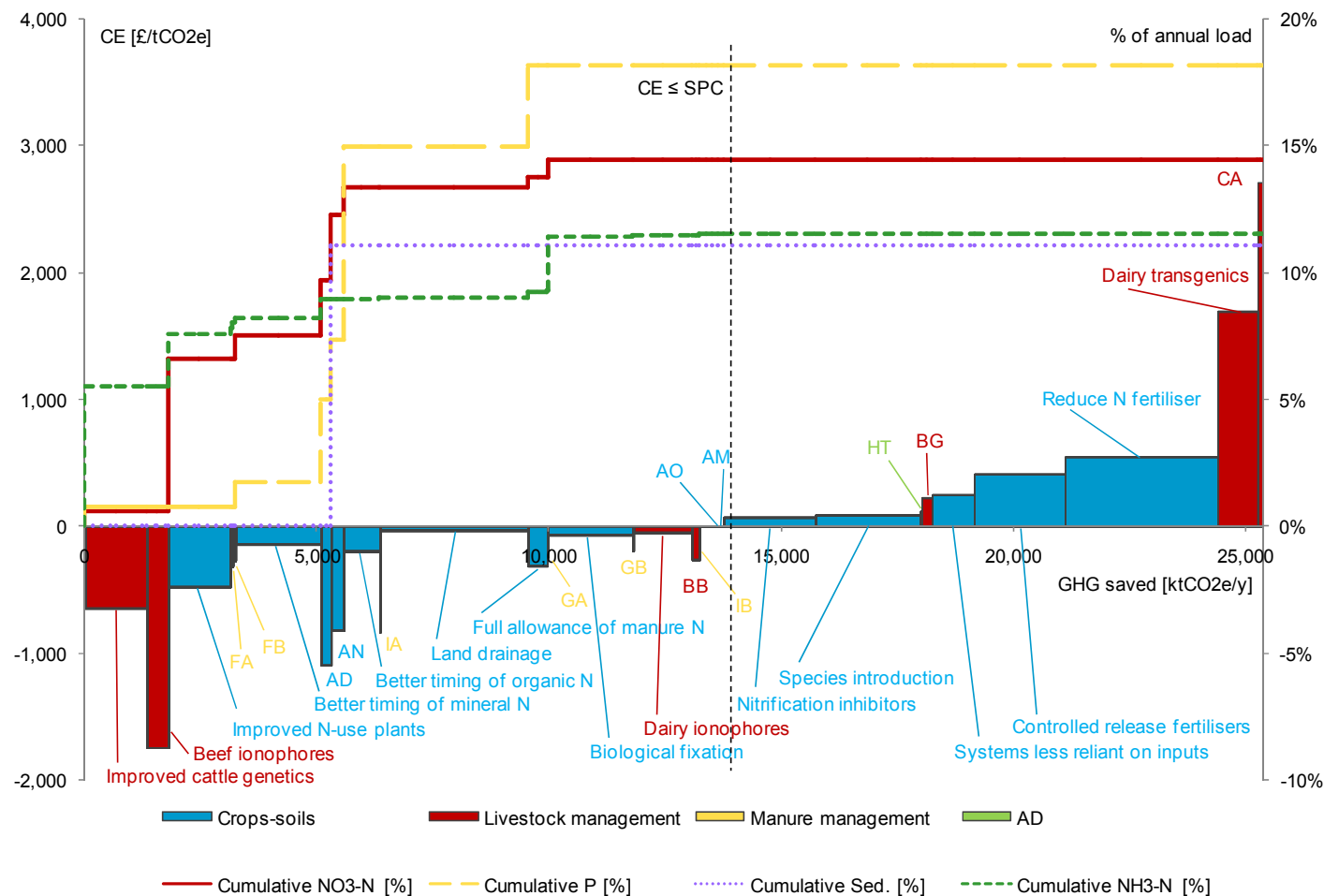


Figure 8. MP-MACC-E

The bars represent the MP-MACC while the lines represent the cumulative savings in the annual load of the four pollutants. Sed.: sediment. See Table 5 for the names of the measures with abatement potential less than 400 kt CO<sub>2</sub>e y<sup>-1</sup>



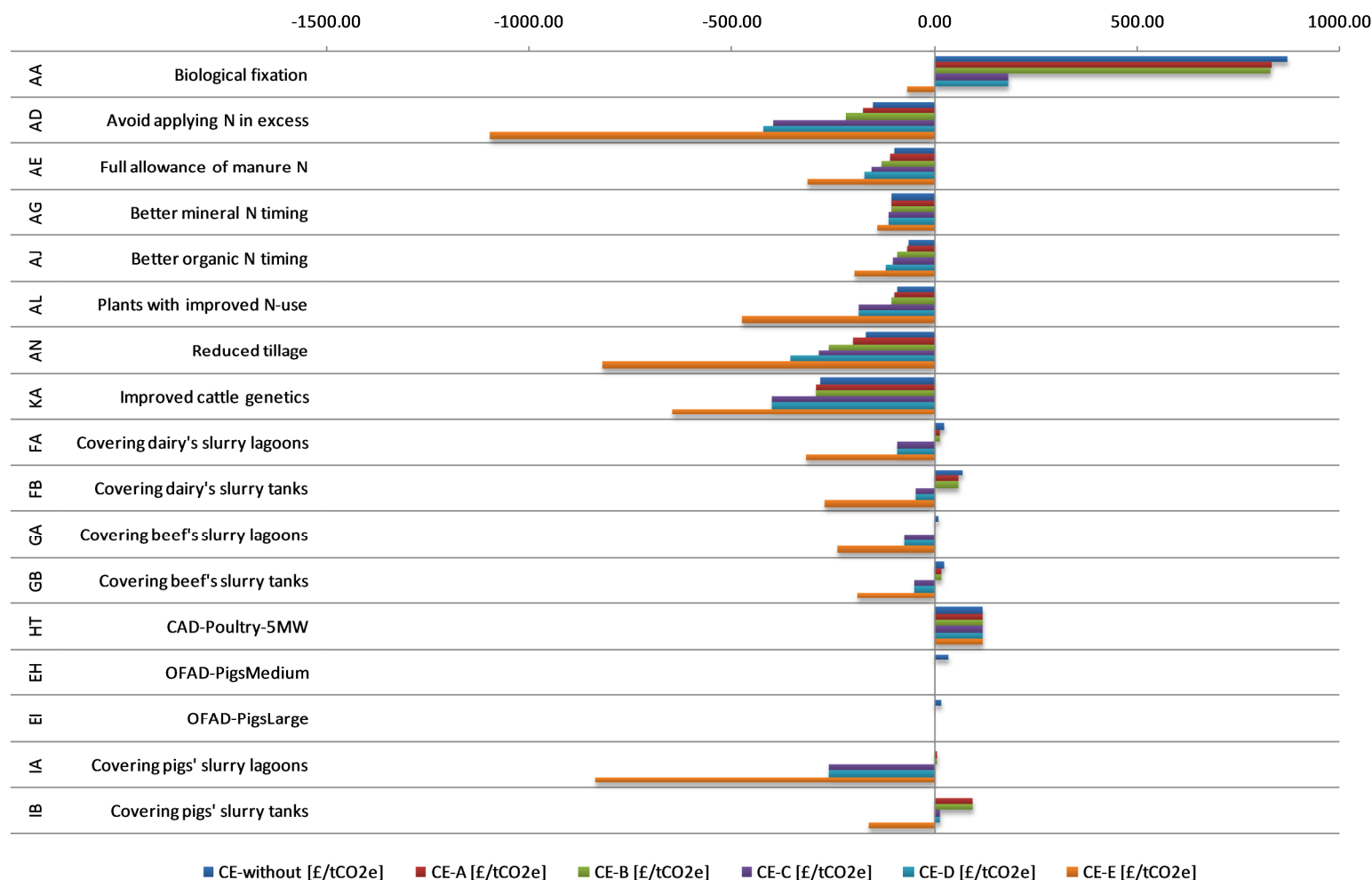


Figure 9. CE of measures on the MACCs which have data on co-effects

CE-without, CE-A, CE-B, CE-C, CE-D and CE-E represents CE values as calculated in GHG-MACC, MP-MACC-A, MP-MACC-B, MP-MACC-C, MP-MACC-D and MP-MACC-E, respectively

## 3.6 Discussion

The results suggest that data on external effects can modify the MACC, but that the inclusion of external effects has surprisingly little impact on the cumulative pollution reductions in this analysis. Fully implementing cost-effective MM in England and Wales could save 37% of GHG emissions annually if no co-effects are included. MP-MACC-A to MP-MACC-D show the same GHG savings and 9-18% saving of the other four pollutants. Applying high damage costs (set E) drives up GHG, NO<sub>3</sub><sup>-</sup> and NH<sub>3</sub> savings by 6%, 1% and 2%, respectively. As Table 8 shows, the overall effect on pollutant abatement is not greatly affected by the different damage costs considered in this study.

*Table 8. Pollution savings from GHG measures with CE < SPC in 2022 in England and Wales expressed as % of BAU agricultural load*

MACC	GHG	NO <sub>3</sub> <sup>-</sup>	NH <sub>3</sub>	P	Sediment	Annual cost (£ m) <sup>1</sup>
GHG-MACC	36.7%	n/a	n/a	n/a	n/a	-1,841
MP-MACC-A	36.4%	13.8%	9.1%	18.1%	11.1%	-1,845
MP-MACC-B	36.4%	13.8%	9.1%	18.1%	11.1%	-1,845
MP-MACC-C	36.6%	13.8%	9.3%	18.1%	11.1%	-1,836
MP-MACC-D	36.6%	13.8%	9.3%	18.1%	11.1%	-1,836
MP-MACC-E	42.2%	14.4%	11.5%	18.1%	11.1%	-1,685
BAU annual load	32.6 Mt CO <sub>2</sub> e	276 kt NO <sub>3</sub> <sup>-</sup> -N	158 kt NH <sub>3</sub> -N	4.0 kt P	1,790 kt sediment	

<sup>1</sup> Negative cost implies saving

There are two main reasons for this. Comparing the monetary value of total pollution loads from agriculture (Table 9), it appears that with the conservative damage cost sets A and B the total negative impacts of non-GHG pollutants are substantially lower than of GHGs. Consequently, measures designed to reduce GHG emissions are likely to have much lower monetary impacts on non-GHG pollution than on GHG pollution. This is reflected on Figure 6, where the monetary value of the MMs' co-effects with damage cost sets A and B is generally 3-10 times lower than the monetary value of GHG savings.

Table 9. Value of agricultural pollution load of GHG,  $\text{NO}_3^-$ ,  $\text{NH}_3$ , P and sediment in 2020 in England and Wales, £ million

Damage cost set	GHG	$\text{NO}_3^-$	$\text{NH}_3$	P	Sediment	Total
A	1,118	60	285	39	44	1,546
B		185	285	181	193	1,962
C		1,184	2,800	385	44	5,531
D		1,184	2,800	181	193	5,475
E		5,681	8,235	181	193	15,408

Based on (Anthony et al. 2008). GHG is valued at  $\text{£}34.3 \text{ t CO}_2\text{e}^{-1}$  (Price et al. 2007)

Increasing the damage costs (set C) makes a difference and Table 9 shows that the value of both the nitrate and the ammonia total agricultural pollution is higher than the value of total agricultural GHG pollution. In parallel, for many MMs the combined monetary values of co-effects are now 2-6 higher than the monetary value of the GHG savings (Figure 6). This should bring those measures with private CE slightly above the SPC, below this threshold. And it does (see measures “Covering pigs’ slurry tanks” and “Covering dairy cattle’s slurry tanks” on Figure 9), but this is not the case for all the measures.

The second reason is the lack of data on external effects for many measures. Only three of the eleven measures for which  $\text{CE} > \text{SPC}$  on the GHG-MACC have data on co-effects, and only one of these has abatement potential considerable enough to change the shape of the MACC (“Using biological fixation to provide N inputs”, with abatement potential of  $1.8 \text{ Mt CO}_2\text{e y}^{-1}$ ). However, this measure becomes efficient only when damage cost set E is applied.

The eight measures for which  $\text{CE} > \text{SPC}$  on the GHG-MACC and external data were not available can potentially change the economically optimal abatement level. Their cumulative annual GHG abatement potential is  $11.6 \text{ Mt CO}_2\text{e}$  – almost as much as the economically optimal abatement level on the GHG-MACC. It is possible that MMs “Nitrification inhibitors” and “Species introduction” would become cost-effective by including their co-effects on nitrate leaching calculated by damage cost set A and B, because their private CE is close to the threshold ( $\text{£}59$  and  $\text{£}69 \text{ t CO}_2\text{e}^{-1}$ , respectively). This change would add further  $4.2 \text{ Mt CO}_2\text{e y}^{-1}$  to the GHG abatement potential under SPC. Damage cost set C, D or E might increase the economically optimal abatement potential by an additional  $6.4 \text{ Mt CO}_2\text{e y}^{-1}$  due to the potentially reduced CE of “Reduce N fertiliser”, “Adopting systems less reliant on input” and “Controlled release fertilisers”. The extremely high CE of the measure

“Transgenic manipulation of ruminants, dairy” might prevent it being a cost-effective measure even with the highest damage cost set applied.

On the other hand, there are some measures currently with no data on co-effects which might have negative effects on one or more of the other four pollutants. For example, “Use composts, straw-based manures in preference to slurry” might increase NH<sub>3</sub> emissions from housing and storage (Chadwick *et al.* 2011, Jungbluth *et al.* 2001).

At least two further caveats should be noted relating to the costs included in this study. The first is that the current work attempts to add only four co-effects to the GHG MACC. It is clear that MMs can have numerous other environmental impacts that should be taken into account. For example, reduced tillage can potentially increase pesticide, fungicide and insecticide use, affecting biodiversity and water quality. Cattle feeding measures requiring more grain could cause an expansion in the area of arable land, with the land use change having negative implications on biodiversity and beyond farm-gate GHG emissions. Reducing N fertilisation below the economic optimum would, again, provoke land use change. The issues of displaced production and full life-cycle costing of MMs are further critiques of existing MACCs, which we have not addressed in this paper.

The second cost issue concerns the avoided control costs related to the reduced non GHG emissions. Implementing a GHG measure might alter the total cost of other pollution control, and this change could be apportioned to GHG measures. In an ideal world, policy would be informed by pollutant-specific MACCs that encompass all relevant cost (including control costs) and benefits. These would have clearly delineated cost boundaries around measures and there would be agreement on where control costs lie. Since in practice the quantification of these costs would require complete knowledge about the other MACCs, and the apportionment might be highly debatable, the current study merely an attempt to apportion external cost impacts (related to GHG measures) to a GHG MACC, without adding the control cost implications the implemented GHG measures have on other pollutants. Because of the multiple effects of single measures, the total control costs of different pollutants cannot simply be added up without the risk of double counting, but should be calculated by adding up the costs of all the measures one by one.

In addition to the biophysical uncertainties, the current study highlights further data needs in terms of the unit damage costs. While the difference between the lower and the higher carbon value estimates used in UK policy appraisal is three-fold, greater differences are revealed in existing estimates for the other damage costs. The complexity of agricultural

externalities and the use of benefits transfer in valuation enhance uncertainties in representing the damage functions for these pollutants. For a discussion of the difficulties associated with environmental valuation, see Smart *et al.* (2011). These uncertainties emphasise the importance of using CE and cost-benefit calculations as a complementary rather than exclusive policy tool, with a clear understanding of their advantages and limitations.

Finally, note that the choice of the carbon threshold will inevitably influence the estimated economically optimal abatement potential. We used the SPC estimated by Price *et al.* (2007), while the UK Government current approach (based on reduction targets) values carbon in the non-traded sector in 2022 at £31 to £93 t CO<sub>2</sub>e<sup>-1</sup>, with a central value of £62 t CO<sub>2</sub>e<sup>-1</sup> (DECC 2009). Using this central value would move the “Nitrification inhibitors” measure into the efficient category on the GHG-MACC.

### 3.7 Conclusion

The omission of external effects has been highlighted as a drawback of GHG MACC analysis in policy making. The evidence presented here shows how the inclusion of external effects can alter the CE of environmental measures and how alternative damage cost estimates for NO<sub>3</sub><sup>-</sup>, NH<sub>3</sub> P and sediment can change the results of abatement potential estimates derived in the 2022 MTP Optimistic GHG MACC. Higher damage cost values (sets C and D) make some measures more cost-effective, improving both the cumulative GHG abatement potential and the associated gains in the other four pollutants. Very high damage costs (set E) would justify the implementation of almost all the GHG measures which have positive co-effects. This finding is in line with the estimated costs of different pollutants originating from agriculture: using lower damage costs (set A and B) the total cost is dominated by GHG emissions, increasing the non-GHG pollutants’ damage costs to set E shrinks to GHG’s contribution to costs to the tenth of its original share.

This study highlights the gaps in data availability for other externalities relevant to GHG MMs. Ongoing and future experimental and modelling research should focus on expanding the scope of research beyond GHG effects, especially in relation to the MMs with high abatement potential, like nitrification inhibitors, controlled release fertilisers, species introduction and systems less reliant on input. Advances in monetary valuation of pollutants are also desired. Notwithstanding the data gaps, the MP-MACC is a useful analytical device

for cumulating knowledge about the GHG mitigation efficiency, co-effects and private costs of GHG MMs, and offers easy visual representation of the integrated information.

The multiple pollutant MACC can offer specific policy messages for agencies trying to interpret MACC information. The first is to focus any further analysis on GHG MMs that are slightly above the threshold on the GHG MACCs, as they most probably have co-effects which could make their implementation worthwhile. The second message is to explore thoroughly any possible negative external effects of those GHG measures that are cost-effective on the GHG MACCs and become cost-effective on the MP MACCs. In these cases it may be useful to consider effects beyond those analysed in this paper, like biodiversity, soil quality, human health, animal health and welfare and social effects (e.g. food security, resilience of rural communities).



## 4 Paper III. Evaluating the cost-effectiveness of agricultural mitigation policy

Vera Eory<sup>1</sup>, Emma Close<sup>2</sup>, Dominic Moran<sup>1</sup>

<sup>1</sup> Research Division, SRUC, West Mains Road, Edinburgh EH9 3JG, UK

<sup>2</sup> Rural and Environment Science and Analytical Services (RESAS), Scottish Government, Saughton House, Broomhouse Drive, Edinburgh, EH11 3XD, UK

### 4.1 Abstract

Scotland has adopted arguably the most ambitious statutory targets to reduce GHG emissions and all sectors of the economy are expected to contribute to this ambition. An economically rational assessment of MMs to reach the target requires the analysis of the relative cost-effective MMs and their potential accommodation within current or potential policies. This paper outlines the difference between technical and policy mitigation potentials in the context of Scottish agriculture, a sector responsible for around a fifth of Scotland's GHG emissions. Technical or maximum feasible potential is informed by original cost-effectiveness analysis of MMs represented in an engineering MACC derived for UK agriculture. The paper draws on the data from a region-specific representation for Scotland and highlights the current shortfall between the full technical mitigation potential shown in a Scottish MACC, and the cost-effective potential currently achievable under existing regional agricultural policies and initiatives, as derived from participatory research. The paper speculates about future policy options for increasing agriculture's contribution in line with the ambition set out in the Climate Change (Scotland) Act 2009.

### 4.2 Introduction

Under the Climate Change (Scotland) Act 2009 Scotland has used its devolved powers to set out ambitious statutory targets to reduce total GHG emissions by 80% by 2050 from a 1990 baseline, with an interim target of 42% by the end of the third carbon budgeting period in 2022. To help ensure the delivery of these globally ambitious targets, the Act requires that Scottish ministers set annual targets, in secondary legislation, for Scottish emissions and ministers must produce a report setting out proposals and policies (the Report on Proposals



and Policies, RPPs) for meeting those targets, and describing how they contribute to the interim and 2050 targets. All sectors of the economy are expected to contribute to this commitment (though not to an equal level). The Scottish Government originally drew on initial CE analysis conducted at the UK level by the Committee on Climate Change to identify efficient MMs available in the economic sectors (e.g. energy production, transportation, agriculture and land use change), accounting for mitigation potential which is only partly reflected in the national inventories in the IPCC GHG accounting system. The Scottish Government's first and second Reports on Proposals and Policies (Scottish Government 2011a, Scottish Government 2013a) attempt to add up this identifiable abatement potential. The documents match sector MMs with existing sector policies and thereby identify potential sector-specific abatement. This exercise demonstrates the challenges in matching currently identified abatement potential with policy ambitions. It highlights the need to ensure all policies, and proposals for future policies, are implemented if Scotland is to meet its statutory targets.

This paper details the participatory development and analysis of the Reports on Proposals and Policies in the Scottish agricultural sector and, more specifically, the associated challenges in translating ambitious mitigation potential from MACC analysis into cost-effective policy. There have been several critiques of MACC analysis highlighting methodological shortcomings and some of the key uncertainties inherent in the approach (DeAngelo *et al.* 2006, Kesicki and Ekins 2012, Moran *et al.* 2011b). The aim of this paper is not to reflect on these elements, although they remain highly relevant to the credibility of policy design. But there appears to be no studies demonstrating the limitations in how the analysis might actually be used in ex ante evaluation of PIs conducted in collaboration with policy makers. The contribution of this paper therefore lies in highlighting the difference between technical and likely policy mitigation potentials. It draws on analysis for the agricultural sector, which presents challenges in terms of measure uncertainty and the extent of available policies to accommodate available MMs. The next section provides background on agricultural emissions and the current opportunities for accommodating mitigation within existing sector policies in Scotland. Section 4.4 details the engineering MACC analysis for agriculture and its link to the determination of cost-effective policy. Section 4.5 outlines the analytical framework and the way UK level data were modified to provide an estimate of efficient abatement potential for Scotland and to include policy considerations when developing the policy abatement potential. Section 4.6 presents the results that contrast technical and policy abatement potentials; section 4.7 provides a concluding discussion.

## 4.3 Agricultural policy and GHG mitigation

Agriculture has been estimated to account for approximately 11-12% of global anthropogenic emissions (Smith *et al.* 2007), and 10% of European anthropogenic emissions (European Environment Agency 2014). Using the same accounting, agriculture's share of GHG emissions in Scotland is 15% (Thistlethwait *et al.* 2012), however, the methodology adopted by the Scottish Government allocates the emissions arising from agriculture-related land use change to the sector, making its share in emissions to 19% (10.5 Mt CO<sub>2</sub>e) (Scottish Government 2013b). This level of importance in GHG emissions might normally be sufficient to see the sector included in formal cap and trade regulation, but agriculture presents specific challenges in terms of measuring and monitoring emissions sources. Specifically, regional variation and the biophysical heterogeneity of farming systems affect emissions, mitigation potential and costs of known mitigation technologies – in terms of IPCC calculations, the regional variation and biophysical heterogeneity is mainly excluded. These differences are further affected by uptake uncertainties reflecting behavioural and attitudinal differences among farmers. Agricultural mitigation policy is further complicated by the extent to which measures can be matched to a limited set of rural climate change PIs, which, for Scotland, are partly determined at the EU level and specifically the limited measures allowable under the Common Agricultural Policy Rural Development Regulations that govern how farm support can be allocated.

Feasible agricultural mitigation potential can be grouped into three broad categories: reducing N<sub>2</sub>O and CH<sub>4</sub> emissions by improved farm efficiency and specific technologies; replacing fossil fuel emissions with alternative energy sources; and enhancing the removal of atmospheric CO<sub>2</sub> via net sequestration into soil and vegetation sinks. These categories can be further sub-divided into a variety of measures with greater or lesser applicability in specific biophysical conditions. For example, in a MACC analysis for the UK Moran *et al.* (2011b) depicted the CE of around 30 measures drawn from a longer list that included measures with small abatement potential or with significant overlapping interactions. But in the absence of relevant policy incentives, not all measures are likely to be implemented even where they are technically feasible.

A review of relevant agricultural policy in Scotland reveals that measures can be accommodated within three identifiable PIs that can directly or indirectly promote agricultural GHG mitigation. First, mandatory Cross Compliance is used to link direct CAP support payments to compliance by farmers with basic environment environmental good

practice (so-called Pillar One payments). Though no direct GHG MM is included in the Cross Compliance regulations, they do currently require compliance with Nitrate Vulnerable Zones regulations. Here, storage and application of inorganic and organic nitrogen fertilisers are more strictly controlled, leading to GHG co-benefits arising from management measures (avoiding the excess use of N fertilisers and using the full allowance of manure N-content). Recent renegotiation of the CAP resulted in the removal of member states' discretion to link compliance of additional national PIs to Pillar One payments, therefore the development of an alternative national mandatory framework can be considered if uptake is not secured on a voluntary basis.

Second, The Scotland Rural Development Programme is the discretionary application of CAP Pillar Two funds for implementing economic, environmental and social measures. The 2014-2020 CAP Pillar Two establishes 'the shift towards a low carbon and climate resilient economy in the agriculture and food sectors and the forestry sector'<sup>1</sup>, and requires that 25% of Rural Development Programme (RDP) funding is allocated to climate change mitigation and adaptation action. As such actions have been rarely featured in RDPs, currently the European Commission and governments of member states are working on mainstreaming climate action in the Programmes. As for Scotland the pre-2014 Scotland Rural Development Programme (SRDP) allowed financial support for some actions that lead to GHG co-benefits; specifically covering slurry stores and anaerobic digestion of animal waste. The new proposal sets out continued support for manure storage measures and efficiency improvement in the beef sector, with further details about available support to be revealed after the European Commission's approval of the Scottish programme (Scottish Government 2014).

As a third PI, the Scottish Government is also funding an additional farm advisory element Farming for a Better Climate (FFBC), specifically designed to promote uptake of various GHG MMs, mainly through improving farm efficiency. FFBC is novel in the EU, and its effectiveness depends on the way measures demonstrated on showcase farms are subsequently adopted by the farming community. As yet the measure has not been evaluated for its overall contribution to the mitigation target.

The RPPs sought to define the sector mitigation potential by aggregating the contribution of measures that could be accommodated by these policies. To do this, CE analysis using

---

<sup>1</sup> COM (European Commission) (2011) Proposal for a Regulation of the European Parliament and the Council on support for rural development by the European Agricultural Fund for Rural Development, COM(2011) 627 final/2 (19 October 2011)

MACCs derived for the UK was transposed onto Scotland, with allowances for regional differences associated with biophysical conditions and the spatial applicability of measures.

## 4.4 MACC analysis

An engineering MACC orders MMs by their GHG CE; i.e. costs per unit of CO<sub>2</sub> equivalent abated. Measures are additional to mitigation activity in a BAU baseline without policy interventions. Measures that have lower CE than a carbon price threshold (representing the marginal benefit of mitigation) help to define an efficient abatement potential for a sector (Figure 10). Engineering MACCs for UK agriculture have been developed and refined (MacLeod *et al.* 2010b, Moran *et al.* 2008); the latter estimating the agricultural GHG abatement potential for Scotland in 2022, under the CE threshold of £72 t CO<sub>2</sub>e<sup>-1</sup> (which is the central carbon value in the non-traded sector in 2022, in 2011 prices (DECC 2009)), to be between 1.2 and 2.5 Mt CO<sub>2</sub>e, assuming full implementation of cost-effective MMs. The most cost-effective MMs identified by MacLeod *et al.* (2010b) include more efficient use of nitrogen, selective livestock breeding, altered livestock feeding practices and anaerobic digestion.

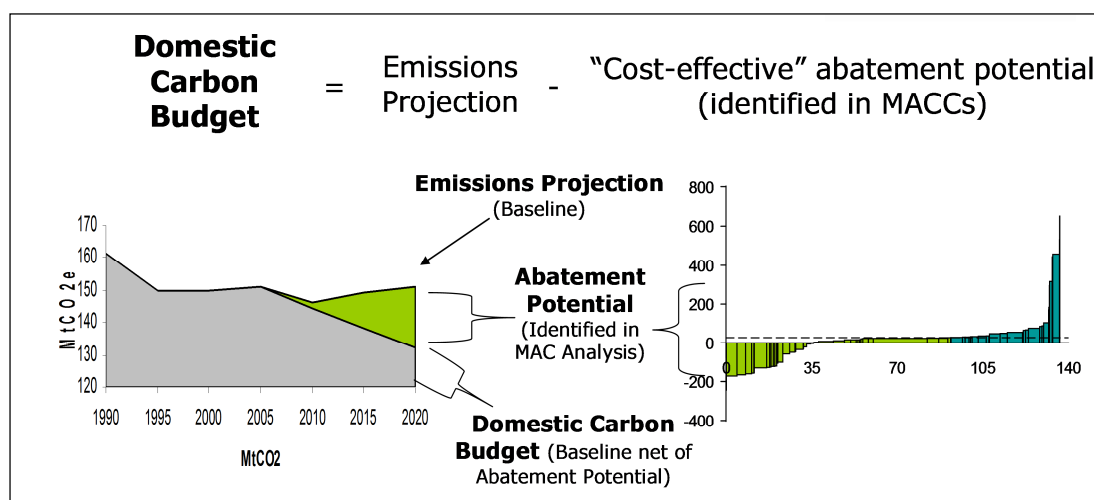


Figure 10. Hypothetical engineering MACC (right) and the corresponding abatement potential deducted from an emissions baseline (left)

Source: (Moran *et al.* 2008)

While MACCs have galvanised debate in UK agriculture, their policy relevance has largely been restricted to informing a process of negotiation between government and industry, leading to the development of an industry road map, the Greenhouse Gas Action Plan of the Agriculture Industry in England (ADAS *et al.* 2012) on voluntary uncompensated MMs. But

there is a lack of analysis of how abatement potential might be delivered through alternative PIs, including the wider options offered by the scope of the CAP. This wider policy space offers a range of mechanisms including advisory/institutional instruments, which are voluntary, providing no financial incentives, mandatory regulatory instruments also without financial incentives, and voluntary or mandatory economic instruments with financial incentives (Pretty *et al.* 2001).

## 4.5 Methods and data

Estimating the CE of the potential policies entailed five analytical stages, all of which were conducted in collaboration with policy stakeholders at the Scottish Government. First the available PIs were clarified. Second, a Scottish MACC were developed from the UK MACC (MacLeod *et al.* 2010b) and the MMs included in it were screened to identify those that could be accommodated within available policies. Third, the MMs were mapped onto the available PIs and the part of the transactions costs of PI implementation (specifically the public sector's policy operation costs, as defined in (Krutilla 2011)) was estimated. Finally the CE of each PI was calculated.

Harris *et al.* (2009) suggest a list of PIs applicable to GHG mitigation effort in the UK agriculture. Partly based on that report, the Scottish Government developed a short list of three currently available PI's. First, mandatory legislation, either at the Scotland or EU level, could potentially be extended to include GHG mitigation requirements that are currently not mandatory (e.g. including in Cross Compliance requirements). Second, the SRDP could be extended to include assistance for measures requiring capital investment, thereby supporting investment in measures such as anaerobic digestion. Third, FFBC could be continued to increase awareness and uptake of low-cost MMs. As noted these PIs suggest different levels of compulsory versus voluntary compliance; hence policy costs and likely uptake rates should be expected to differ.

The UK MACC (MacLeod *et al.* 2010b) was modified to represent Scottish agriculture better; the Scottish Government led a stakeholder consultation with the Agriculture and Climate Change Stakeholder group to review the abatement and applicability parameters used in the UK MACC. The MMs included in the resulting Scottish MACC were then screened according to four criteria:

1. CE in the Scottish MACC is below to a carbon price forecasted to be £72 t CO<sub>2</sub>e<sup>-1</sup> in 2022 (DECC 2009).
2. Robustness of estimated abatement potential.
3. Public and farmer acceptability; as perceived by the policy stakeholders.
4. Legal status of the measure; i.e. whether the measure is currently allowed under national or EU legislation.

This reduced 29 feasible measures to a short list of 12 (Table 10). Eight measures were not cost-effective, one measure had high mitigation uncertainty ('Improving land drainage'), three were considered to be unacceptable to farmers and the public ('Adding propionate precursors to the diet', 'Adding probiotics to the diet' and 'Transgenic manipulation of animals'), two are illegal in the EU ('Adding ionophores to the diet' and 'Administering bovine somatotrophin to animals'), two were excluded due to policy conflict; specifically 'Covering slurry tanks' and 'Covering slurry lagoons' were not promoted because they are incompatible with anaerobic digestion of manure and policy makers felt that this would be an obstacle in delivering clear messages to farmers. One measure, 'Centralised anaerobic digestion of manure', was excluded as being beyond the scope of individual farm-level decision making.

Table 10. CE of MMs in the 2022 Scottish MACC and allocation to PIs

Mitigation measure	CE in 2022 MACC <sup>1</sup>	Included in policy?	Reason for exclusion	Policy instruments		
				Potential mandatory legislation	FFBC	SRDP
Using biological fixation to provide N inputs (clover)	CE > C value	No	Not cost-effective			
Reducing N fertiliser	CE > C value	No	Not cost-effective			
Improving land drainage	0 < CE < C value	No	Uncertainty on overall GHG and nitrogen leaching effects			
Avoiding N application in excess	CE ≤ 0	✓		✓	✓	No
Using manure N to its full extent	0 < CE < C value	✓		✓	✓	No
Introducing new species (including legumes)	CE > C value	No	Not cost-effective			
Improving the timing of mineral N application	CE ≤ 0	✓		✓	✓	No
Using controlled release fertilisers	CE > C value	No	Not cost-effective			
Using nitrification inhibitors	CE > C value	No	Not cost-effective			
Improving the timing of slurry and poultry manure application	CE ≤ 0	✓		✓	✓	No
Adopting systems less reliant on inputs	CE > C value	No	Not cost-effective			
Adopting plant varieties with improved N-use efficiency	CE ≤ 0	✓		No	✓	No
Separating slurry applications from fertiliser applications by several days	CE ≤ 0	✓		✓	✓	No

<sup>1</sup> C value: carbon value in 2022 (DECC 2009)

Table 10. cont.

Mitigation measure	CE in 2022 MACC <sup>1</sup>	Included in policy?	Reason for exclusion	Policy instruments		
				Potential mandatory legislation	FFBC	SRDP
Using reduced tillage and no-till techniques	$CE \leq 0$	No	Not cost-effective			
Using composts, straw-based manures in preference to slurry	$CE \leq 0$	✓		✓	✓	No
Increasing the starch concentrate in diet – dairy/beef	$CE > C$ value	No	Not cost-effective			
Increasing maize silage in diet – dairy	$CE \leq 0$	✓		No	✓	No
Adding propionate precursors to the diet – dairy/beef	$CE \leq 0$	No	Not acceptable for farmers			
Adding probiotics to the diet – dairy/beef	$CE \leq 0$	✓		No	✓	No
Adding ionophores to the diet – dairy/beef	NA	No	Illegal			
Administering bovine somatotropin – dairy	$CE > C$ value	No	Illegal			
Transgenic manipulation of animals – dairy	$CE > C$ value	No	Not acceptable for the general public			
Improved genetic potential of animals – dairy cows' fertility	$CE \leq 0$	✓		No	✓	No
Improved genetic potential of animals – dairy cows' productivity	$CE \leq 0$	✓		No	✓	No
Improved genetic potential of animals – beef cattle	$CE \leq 0$	✓		No	✓	No

<sup>1</sup> C value: carbon value in 2022 (DECC 2009)



Table 10. cont.

Mitigation measure	CE in 2022 MACC <sup>1</sup>	Included in policy?	Reason for exclusion	Policy instruments		
				Potential mandatory legislation	FFBC	SRDP
On-farm anaerobic digestion of manure – dairy/beef/pigs	Pigs/beef: $0 < CE < C$ value; Dairy: $CE > C$ value	✓		No	No	✓
Centralised anaerobic digestion of manure – dairy/beef/pigs/poultry	Pigs: $0 < CE < C$ value; Dairy/beef/poultry: $CE > C$ value	No	Not deliverable at individual farm level			
Covering slurry lagoons – dairy/beef/pigs	$0 < CE < C$ value	No	To avoid a confusing message (incompatible with anaerobic digestion)			
Covering slurry tanks – dairy/beef/pigs	$CE > C$ value	No	To avoid a confusing message (incompatible with anaerobic digestion)			

<sup>1</sup> C value: carbon value in 2022 (DECC 2009)

Stage three matched MMs to the available PIs under their current operation, and as they may evolve over the carbon budgeting period. This matching was provided by policy stakeholders. All measures except 'On-farm anaerobic digestion of manure' were assigned to FFBC. Anaerobic digestion (AD) was considered too capital intensive to be included in this group and is therefore included under a financial provision within SRDP. A subset of FFBC measures were considered likely to transition from voluntary to mandatory from 2018 if voluntary uptake does not reach 90% by that year (see in Table 10: 'Potential mandatory legislation'). The form of mandatory PI was not specified at this stage of the policy process.

In the current analysis policy transaction costs were partially estimated, specifically the public policy operation costs, not considering policy formulation and implementation costs and transaction costs borne by private agents. In the environmental policy literature transaction costs are contested with few reliable estimates available covering land-based GHG mitigation (Krutilla 2011, Rorstad *et al.* 2007). Given the lack of estimates to other elements of the transaction costs, only the public policy operation costs were quantified. These cost assumptions were developed by policy stakeholders to reflect their expectations on the budgetary requirements of future policy delivery. The costs of extending mandatory legislation to include GHG measures were assumed to be negligible as it was envisaged that it would be tied to existing CAP legislation (Table 11), therefore no additional costs were allocated to this PI. Those for FFBC were set equal to the historic costs of this GHG PI scheme, assuming that expanding the advisory activities to cover more MMs would not generate additional costs, but at the same time allocating the cost of the scheme to GHG mitigation delivery. Uptake of anaerobic digestion measures is supported by 50% of the capital investment; this transfer payment was taken as the public policy operation costs for this PI, while the private cost of the measure was reduced with the same amount. The assumed uptake and policy coverage of PIs (i.e. number of farms) was based on estimates suggested by Harris *et al.* (2009) and on consultations with Scottish Government (Table 11). For example 90% of land in Scotland is estimated to be under Pillar One provisions, so the coverage of a mandatory policy that would be tied to Pillar One is 0.9. This is a conservative estimate, as CAP post 2014 may have higher coverage. Uptake is assumed to be 90%, given the regulatory nature of the PI. FFBC is expected to cover around 50% of farms but is assumed to target the largest emitters, which might account for approximately 90% of total farm GHG emissions. Coverage is therefore 0.9. FFBC is a voluntary PI, and therefore a maximum 50% uptake is assumed for currently available N efficiency measures, and a maximum of 15% uptake is assumed for measures that are less easily implemented, or are

not currently available. SRDP coverage is assumed to be 100% and its uptake is estimated to be 10%, due to the capital investment required for the anaerobic digestion measures.

Accounting for interactions between PIs is important since some policies target the same measures and each measure has a maximum level of applicability. It is important to avoid double-counting of uptake by being clear on the assignment of mitigation and cost arising from additional uptake to the PI that is responsible for the additional uptake.

*Table 11. Uptake, coverage and public policy operation cost assumptions for PIs.*

	Potential mandatory legislation	FFBC	Funding for AD under SRDP
Coverage	0.9	0.9	1
Maximum uptake	0.9	0.5 / 0.15	0.1
Temporal uptake pattern	Maximum from year 1	Linear increase (maximum 2027)	Linear increase (maximum 2027)
Annual public policy operation cost [£m]	0	0.15	50% of private costs

*Finally the CE values of the PIs were calculated annually until 2022. The total Based on (Scottish Government 2011b)*

Finally the CE values of the PIs were calculated annually until 2022. The total annual cost was derived by adding up the annual private costs related to the measures targeted by the PI and the annual public policy operation cost of the PI. The total annual abatement potential arising from the PI was the sum of the annual abatement of the MMs targeted by the PI. Interactions between all the measures were considered to avoid overestimating the abatement potential (MacLeod *et al.* 2010b). In this process the abatement potential of each measure was modified by factors related to the more cost-effective measures, assuming those get implemented on the farm first. The factor, developed from expert judgement, represented the difference between the mitigation achieved if the two measures are applied separately versus if they applied together. Potential interactions between the MMs regarding their costs were not considered in the analysis. Finally, the total annual cost was divided by the total annual abatement potential (Equation 3).

*Equation 3*

$$CE_{PI} = \frac{\sum_{i=1}^m Private\ costs_{PI,i} + Transaction\ cost_{PI}}{\sum_{i=1}^m GHG\ abatement_{PI,i}}$$

Where:

*i*: MM *i* from a set of *m* measures assigned to PI

## 4.6 Results

Table 12 sets out the estimated 2022 CE of the three PIs. The CE of FFBC and mandatory legislation are both negative, as they are mostly comprised of zero-cost or win-win measures, providing financial gains to farmers. The single measure that is not win-win has a CE value close to 0. The SRDP PI's CE is £84 t CO<sub>2</sub>e<sup>-1</sup>, which is above the threshold carbon value of £72 t CO<sub>2</sub>e<sup>-1</sup>. This is because the SRDP targets anaerobic digestion measures on all farm types, most of them having CE > 0, and some of them having CE higher than the carbon price threshold.

*Table 12. CE and abatement potential PIs in 2022*

	Potential mandatory legislation	FFBC	Funding for AD under SRDP
CE [£(2011) t CO <sub>2</sub> e <sup>-1</sup> ]	-95	-189	84
Abatement potential [kt CO <sub>2</sub> e y <sup>-1</sup> ]	227	146	10

The policies planned by the Scottish Government can deliver an estimated 383 kt CO<sub>2</sub>e GHG saving in 2022 in agriculture, which is a 2.7% reduction from 1990 baseline Scottish agricultural emissions (Scottish Government 2013b). Figure 11 illustrates the relative abatement potential highlighting the disparity between current policy potential and the agronomic or full technical potential. The Scottish agricultural MACC for 2022 estimated that 2,584 kt CO<sub>2</sub>e GHG saving would be achievable by implementing all technically feasible measures by all farmers, 1,144 kt CO<sub>2</sub>e saving would be achievable assuming an uptake of the MMs on 45% of the land area and livestock, and 636 kt CO<sub>2</sub>e could be saved by implementing only the cost-effective measures (measures with CE < C value) on 45% on land area and livestock. A 45% uptake corresponds to the assumed coverage reached for incentive-based PIs (Moran *et al.* 2008).

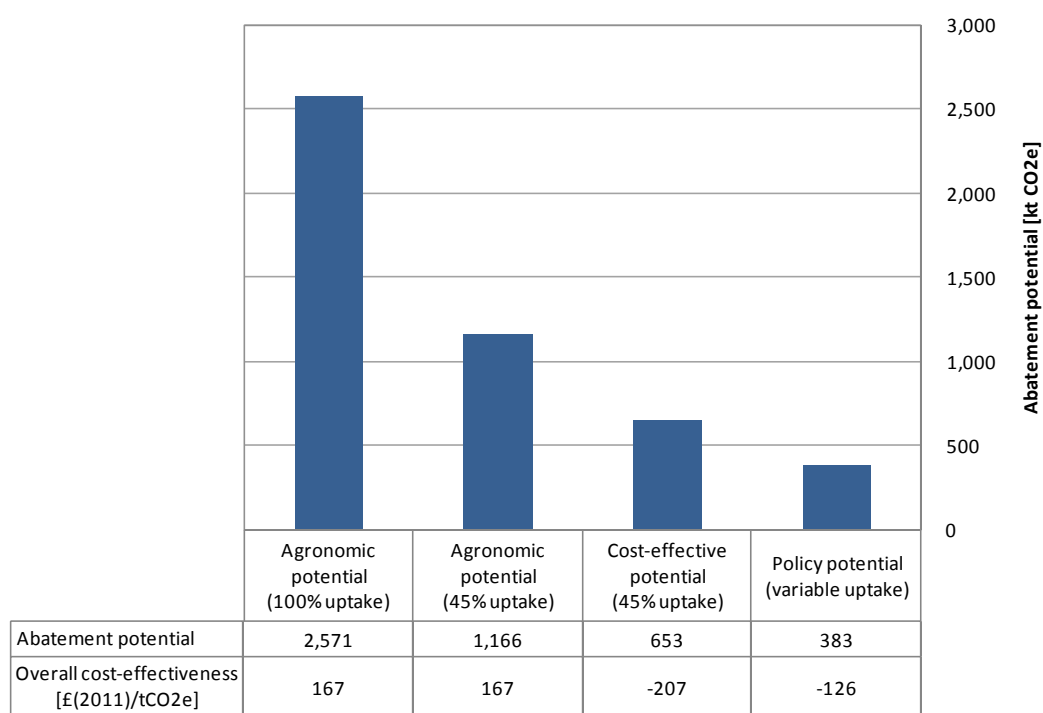


Figure 11. Abatement potential and overall CE of MM packages

*Agronomic potential: achieved by the implementation of all technically feasible measures. Cost-effective potential: achieved by the implementation of measures which are cost-effective relative to a carbon threshold. Policy potential: achieved by implementation of cost-effective measures accommodated by available PIs*

## 4.7 Discussion

Scotland's climate change policy has received considerable international attention, although the challenge of realising the level of ambition is only now becoming clear. Devolved government in Scotland means that UK analysis must be transposed onto a different policy landscape and the level of ambition in the Climate Change (Scotland) Act 2009 increases the urgency to identify CE abatement potential in all sectors, with agriculture and land use change identified as providing considerable low cost and win-win potential.

A report by Audit Scotland (Audit Scotland 2011) calculated the overall costs of meeting the 2020 target at between £10-11 billion and suggested that the Scottish Government had not yet sought to prioritise within its set of policies based on a CE analysis. This paper demonstrates some of the challenges of doing so for the agricultural sector, drawing on a participatory MACC analysis. The sector presents specific challenges that have been outlined in a previous MACC study. In this study we considered further adjustments to that analysis and the added hurdle of considering available PIs for accommodating available

MMs. The current policy constraints, as the analysis was informed by policy makers, mean that there is significantly reduced abatement potential when moving from a MACC showing technically feasible and cost-effective potential to one that accommodates measures that are feasible to support via PIs.

The RPPs present aggregate mitigation data by counting the potential contributions of policies and proposals available in all sectors. These contributions are expressed in terms of a policy ambition rather than a definitive set of policy targets. Eventually, in each sector mitigation will be determined by eventual policy choices that will be decided and published by the Scottish Government. This reflects political decisions made by considering different sources of evidence in addition to CE. The way these may or may not add up to meet targets set in the Scotland Act (2008) will be reported on periodically, but there is strong desire to avoid attribution of fixed targets to sectors and to maintain flexibility to accommodate new cost-effective mitigation potential wherever it arises.

Importantly the RPPs mitigation targets are based on expert based estimates of potential measures, derived from technical abatement. This technical abatement can be partially reflected in the national GHG inventory, but the currently used IPCC 1997 and IPCC 2000 Tier 1 and Tier 2 methodologies (IPCC 1997, IPCC 2000) and the IPCC 2006 methodology to be introduced in 2015 (IPCC 2006) leave many MMs fully or partly ‘invisible’ for the inventory (Moran *et al.* 2008). For instance, many measures targeting organic and mineral nitrogen use might result in reduced fertiliser use without impeding agronomic performance – these reductions in N use will directly appear in the inventory. However, these measures might also change the biogeochemical processes in the soil and thus reducing the proportion of nitrogen being lost as N<sub>2</sub>O – this change in emission factors will not be captured by the methodology. Similarly, most of the enteric CH<sub>4</sub> mitigation effects of feed additives for ruminant are not accounted for in the current and incoming inventory. Nevertheless, inventory development work is in progress to enable the proposed MMs to contribute to the mitigation target in the future (see for example the GHG Platform project in the UK: <http://www.ghgplatform.org.uk/Projects/AC0114.aspx>).

From a sustainable intensification viewpoint both the emission reduction potential and the financial implications of the MMs are important factors, along with other effects, like changes in the emissions of other pollutants, impacts on animal welfare, biodiversity, and also importantly, effects on agronomic performance. Four crop and soil MMs assessed in this work are expected to have negative effect on the yield, up to a 30% reduction (‘Using

biological fixation', 'Reducing nitrogen fertilisation', 'Introducing new species' and 'Adopting systems less reliant on inputs'). These measures have all been excluded from the policy mix for various reasons. The rest of the measures have either no implications on the crop or animal productivity or they are increasing the yield, like all the livestock measures apart from the manure management ones. Those included in the policy mix have the potential to improve the sustainability of Scottish agricultural production not only in terms of GHG emissions but also regarding land use and food security.

Agriculture and related land use, like all other sectors, needs to increase the level of GHG abatement and there are national policy choices about the level of voluntary versus mandatory regulation in the sector. UK and Scottish policy is currently based on a preference for voluntary agreement with the farming sector, and the extent of mitigation to be realised by improved advisory and extension services is currently being assessed before other policy options are tabled. Specifically, as recognised with FFBC there is a focus on understanding the drivers of farmer behavioural change so that uptake of cost-effective measures can meet its maximum potential without recourse to legislation.

Beyond this, CAP legislation circumscribes some of the policy choices available in Scotland. As a European region with its own Rural Development Program, Scotland has scope to define some of its own set of agricultural policies within the limitations of European Union rules for Pillar One and Two instruments. Nevertheless, 2014-2020 CAP cross compliance regulations cannot include member state specific regulations, and the current proposal of SRDP operations propose financial support only for manure management and efficiency improvement in the beef sector. However, mandatory regulations, which are stricter than EU regulations, might be implemented by member countries, as, for example, has been the case in Denmark with regard to nitrogen pollution (Kronvang *et al.* 2008). Beyond these developments, the removal of some legal barriers at EU level could offer scope to unlock further potential by improving the efficiency of meat and milk production (e.g. by administering bST or ionophores). But these measures remain controversial in many countries and the Scottish Government has intimated that their implementation is unlikely even if permitted by the European Union. This repudiation of inherently cost-effective technologies will represent a considerable barrier to the realisation of sector ambitions. It also highlights some of the contradictions and trade-offs inherent in agendas for sustainable intensification of agriculture.

Ultimately the penetration of carbon price as a pollution signal in the sector remains conspicuously low. Carbon labelling of agricultural products provides one market-based signal. But more general exposure to carbon trading would arguably represent a highly cost-effective policy option if implemented internationally. Specific sector challenges remain in terms of measuring, monitoring and reporting emissions. But as these are overcome the implementation of agricultural emissions trading may become feasible.





## 5 Paper IV. Assessing uncertainty in the cost-effectiveness of agricultural greenhouse gas mitigation

Vera Eory<sup>1</sup>, Cairistiona F. E. Topp<sup>1</sup>, Adam Butler<sup>2</sup>, Dominic Moran<sup>1</sup>

<sup>1</sup> Research Division, SRUC, West Mains Road, Edinburgh EH9 3JG, UK

<sup>2</sup> Biomathematics & Statistics Scotland, JCMB, The King's Buildings, Edinburgh, EH9 3JZ, UK

### 5.1 Abstract

It is important to highlight uncertainties in data that feed into estimates of mitigation potential, particularly when these are used in policy development. This paper analyses the uncertainty of mitigation estimates provided by a MACC developed initially to quantify cost-effective mitigation potential in UK agriculture, and subsequently used to inform mitigation policy in Scotland. Qualitative assessment identified the different sources and types of uncertainty in the cost-effectiveness analysis of MMs. Quantitative assessment estimated the statistical uncertainty of the results by propagating uncertainty through the MACC via Monte Carlo analysis, assuming low, medium and high uncertainty in the input variables. The results show that the uncertainty around the economically optimal abatement potential on Scottish agricultural land is  $\pm 10\text{-}51\%$ ,  $\pm 22\text{-}77\%$ ,  $\pm 40\text{-}107\%$ , with low, medium or high levels of uncertainty, respectively. But the ranking of the MMs is relatively robust even with a high level of uncertainty; especially for MMs that are cost-effective relative to a carbon price threshold. The results imply that although there is large uncertainty in abatement potential estimates, there is more certainty about which MMs should be implemented on farms.

### 5.2 Introduction

Policies designed to promote climate change mitigation should be informed by sound scientific evidence on the effectiveness of possible MMs (i.e. their GHG abatement potential, feasibility and cost). But this information can be uncertain, and ignoring this uncertainty can result in sub-optimal recommendations and inefficient policy. Robust policies, which aim to achieve their environmental, economic and social objectives across a range of possible

futures, therefore require that these uncertainties are taken into account (Lempert and Schlesinger 2000).

High variability in agricultural GHG emissions across farms significantly constrains robust GHG emission quantification (Olander *et al.* 2013), and is a barrier to the implementation of market-based instruments to their management. Instead policies in the sector largely rely on alternative instruments including information provision, capacity building and voluntary compliance. These approaches either require no emissions monitoring, or monitoring of management (i.e. input) practices, but not the emissions (e.g. statutory limits on nitrogen fertiliser use). While there is some need to inform such policies by farm-level GHG emission quantification, policy makers still require an evidence-based rationale for the choice of management MMs to be supported, and information on uncertainty associated with the effectiveness and costs should be an integral part of the advice.

Uncertainty analysis is integral in key areas of climate science and its policy interface, both in the physical sciences (e.g. climate modelling), and in economics. Peterson (2006) offers an extensive overview of economic models of climate change impacts, and describes how, in addition to uncertainties in emissions scenarios, the models incorporate uncertainties related to future GHG mitigation and/or mitigation costs. Such results are particularly valuable for high level (global, regional) policy decisions, but are limited in advising policy development at the national level, where information on specific MMs and sub-sectors is most urgently required.

To date, research on the economics of GHG mitigation in agriculture has rarely included significant uncertainty analysis. Exceptions are Meyer-Aurich *et al.* (2012), who present an uncertainty analysis of mitigation potential of biogas production in Germany, and Gibbons *et al.* (2006), who provide farm level GHG mitigation potential and cost estimates for a UK farm with uncertainty reported for the total emissions and for one MM. The lack of analysis and reporting of uncertainty in agricultural economic assessments can be partly explained by difficulties imposed by the heterogeneity of the sector (regarding farming systems, practices, climatic and soil conditions and farmer behaviour), and by the variety of implementation methods for MMs, both of which impede the availability of uncertainty information on model inputs.

Information on uncertainty can be accommodated in policy decision making in different ways, and there is a range of decision support tools to help communicate uncertainty to policy makers. Some uncertainty can be captured quantitatively and then directly included in

economic assessments as, for example, in a propagation of uncertainty in economic models (Tol 1999) or in CBA with real options (Maart-Noelck and Musshoff 2013). Other tools, such as robust decision making techniques (Hallegatte *et al.* 2012, Kann and Weyant 2000, Vermeulen *et al.* 2013), acknowledge a higher level of complexity in uncertainties, and offer an empirical way to consider unquantifiable elements of uncertainty. But in both cases the complexity of incorporating uncertainty in the analysis often negatively impacts upon the knowledge exchange between scientist and policy makers, and therefore results in a limited integration of uncertainty information in the decision making process (Knaggard 2013). A mutual engagement from both scientists and policy makers is required to overcome some of the obstacles in communicating and utilising uncertainty information (Smith and Stern 2011).

This paper attempts to account for uncertainty systematically in the context of abatement CE of GHG mitigation in agriculture. The paper revisits data used to derive the GHG MACC developed for UK agriculture (Moran *et al.* 2011b), restricting the analysis to Scottish agricultural land (croplands including temporary grasslands). The work aims to improve MACC analysis that is used to provide policy recommendations for promising MMs in the agricultural sector. The analysis consists of two parts. First establishing an inventory of the uncertainties that influence CE assessments in agricultural GHG mitigation; second quantitatively appraising uncertainty associated with the CE of GHG mitigation with a focus on Scottish agricultural land.

MACCs are decision making tools widely used to estimate the optimal level of mitigation effort in various sectors of the economy, and to prioritise MMs in terms of their CE (i.e. the cost of GHG abatement). MACCs show the cost of reducing pollution by one additional unit as a function of the cumulative pollution reduction, featuring MMs in the order of their CE. MACCs have informed climate change policy in the EU, US and UK (Kesicki and Strachan 2011). Decision makers often face trade-offs between investing additional time into the procurement of additional information and the benefits associated with ‘better’ decisions resulting from such an investment. MACCs’ enable condensed information, including uncertainties, to be conveyed in a relatively simple way. But, while the visual attractiveness of MACCs can facilitate access to rather complex information, this poses the risk that non-graphical information on key assumptions and, of interest for this paper, uncertainty is overlooked. The general absence of an uncertainty analysis in MACCs has been identified as a potential methodological shortcoming (Kesicki and Ekins 2012), particularly for the land use sector. This paper addresses this shortcoming and makes further contributions by

deriving suggestions for augmenting MACC results with information on uncertainties and providing recommendations on how key MACC input data should be prioritised and collected.

The paper is structured as follows. The sources and types of uncertainties in CE assessments are explored in Section 5.3 and 5.4, respectively. The data and methods from a quantitative uncertainty assessment are presented in Section 5.5, followed by a presentation of results in Section 5.6. Section 5.7 discusses the importance of the different sources of uncertainty and examines the quantitative results. The concluding section provides recommendations for policy makers and for future research.

### **5.3 Sources of uncertainty in the economic assessment of agricultural GHG mitigation**

Uncertainties associated with uptake levels, mitigation potential and costs of future GHG mitigation activities are embedded in a complex feedback loop linking the environment and the economy. Figure 12 highlights the interactions between GHG mitigation and policy and a series of uncertainties. The representation of the environmental processes (GHG concentration, weather, systems impact) is dominated by biogeochemical uncertainties, while the representation of economic activity and the societal impacts are associated with uncertainties related to technological solutions, economic processes, human behaviour and politics.

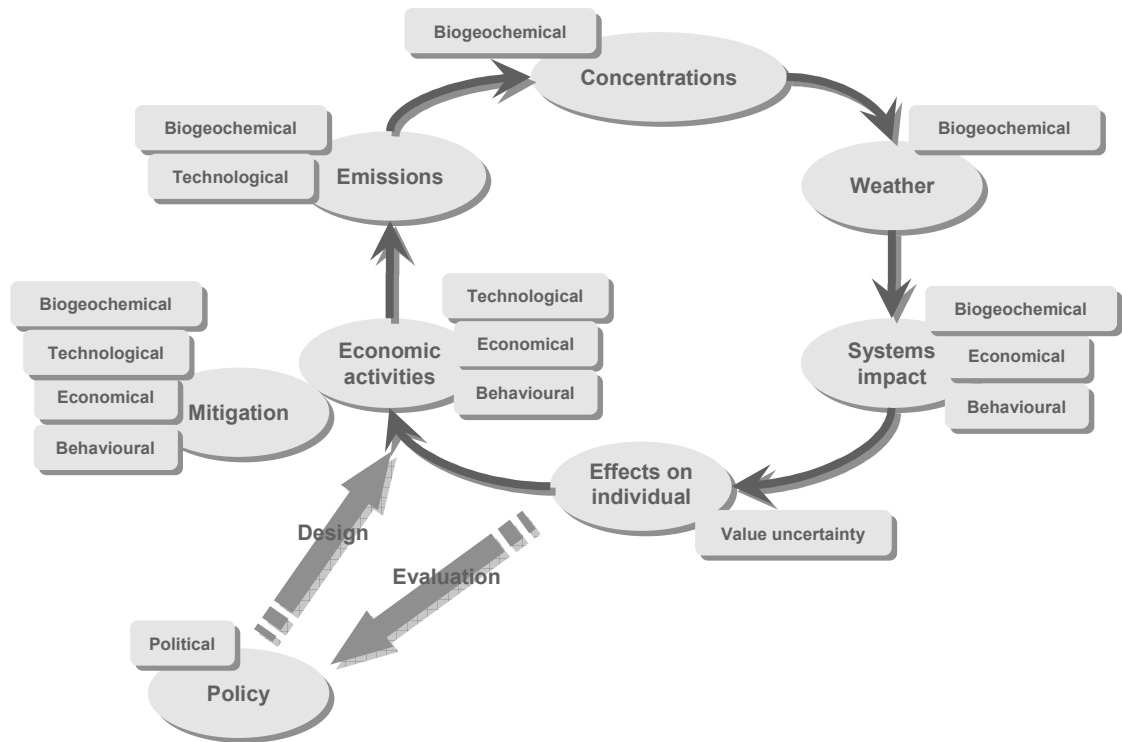


Figure 12. Sources of uncertainty (in squares) in the climate change feedback loop  
Based on Smith and Stern (2011)

In the case of agriculture and land use, biogeochemical processes have a significant influence on both land use activities and associated emissions, and therefore play a key role in determining the cost-effectiveness of MMs. Hence models of land use decisions (e.g. cropping activities, livestock densities, farm management activities) are affected by biogeochemical uncertainties. For example, weather variability induces variability in  $N_2O$  emissions triggered by various biophysical processes, resulting in uncertainty associated with their modelled levels. Weather conditions also impact on farmers' decision-making on timing and amounts of nitrogen fertiliser application, which in turn affects  $N_2O$  emissions and ultimately the actual effectiveness of MMs. The economic and policy environment greatly influences land use decisions and associated agricultural management activities. Therefore, economic and political uncertainties also intervene in model representations. For example, changes in market prices over time, coupled with agricultural and energy policies, will impact both on the land area used for human food and animal feed production relative to other uses (e.g. energy), and on the financial costs and benefits of GHG mitigation technologies. A further uncertainty is related to farmer and other land manager behaviour, which, combined with the policy environment determine the diffusion of mitigation technologies with a direct effect on total GHG abatement. Biogeochemical, economic and

behavioural uncertainty prevails in the impact of climate change on society, translating into an uncertainty in the marginal benefit of mitigation and thus in the carbon price threshold (for a review on the uncertainty of the economic impact of climate change see Tol 2012). Adding to the complexity of the uncertainty problem, the different sources of uncertainties might correlate with each other, imposing further difficulties in uncertainty analysis.

## 5.4 Uncertainty in the economic assessment of agricultural GHG mitigation

Quantifiable uncertainties that can be included in numeric models are referred to as statistical uncertainty (or imprecision, Knightian risk, conditional probability), and can be expressed via probabilities. For example the 100-year global warming potential of N<sub>2</sub>O is estimated to be in the range of 194 - 402 with 90% confidence and a mean estimate of 298 (IPCC 2007). In the agricultural context, statistics about current and historic cropping and livestock activities, input and output prices, experimental data of gaseous emissions and carbon sequestration all have statistical uncertainties, even though this information is not always reported, or not in a form suitable for subsequent economic assessments. Apart from the uncertainties in experimental data, statistical uncertainty is also associated with models, which are imperfect representations of reality. These uncertainties can be quantified if a direct comparison of model outcomes with observed data (i.e. validation) is possible. For example, results of a farm economic model predicting changes in profit can be compared to existing time series data, and the error in the results can be quantified.

Some forms of uncertainty cannot be quantified statistically. So-called deep uncertainty (or ambiguity, Knightian uncertainty) can arise for many reasons, and is particularly relevant to models of complex systems that predict future outcomes (Hallegatte *et al.* 2012, Smith and Stern 2011). A third broad category of uncertainty, value uncertainty occurs if a value depends on personal judgement. Examples include discount rates chosen to reconcile preferences of future and current generations, or the value of human life (Kann and Weyant 2000). Value uncertainty can be illustrated using scenarios to represent the different choice of values. But since probabilities cannot be assigned to the different values, the results of the scenarios cannot be aggregated in the statistical sense.

The qualitative and quantitative assessment in this paper explores uncertainty in cropland and grassland GHG MMs included in a MACC for Scotland. The data are derived from both

biophysical and economic models, complemented by information from expert opinion where observed data and suitable models are unavailable. Because the output of MACC analyses (CE or optimal abatement) cannot be compared to observed data, they can be neither calibrated nor validated. This, in itself, is a deep uncertainty of MACC analyses. Despite this validation challenge, uncertainty information can still be obtained on the results of the MACCs by analysing statistical and deep uncertainties of their inputs. If information on the statistical uncertainty of the inputs is available, the statistical uncertainty can be propagated through the stages of MACC construction.

The main uncertainties associated with the economic assessment of agricultural GHG mitigation are described in Table 13. Deep uncertainties prevail in all of the model inputs. Value uncertainties exist regarding the global warming potential (GWP) metric and the discount rate. In the latter case different rates can be used to reflect private and public decision making. Similarly, in the appropriate policy context, different GWPs can be used in different scenarios, and even though the 100 year time horizon GWP is by far the most widely used, other GWPs could be regarded as more appropriate in certain cases. Deep uncertainties also arise from the underlying modelling processes. This is partly a result of predictions about the future of a complex ecological-economic system under future climate change, and partly related to a lack of information on statistical uncertainty of the underlying models. Uncertainties can, at least in theory, be quantified wherever data are collected about current natural, economic or behavioural phenomena, such as energy prices, current uptake of low-carbon technologies by farmers, or enteric CH<sub>4</sub> emissions from cattle. However, given the spatial and temporal variability in these phenomena, modelling is often needed to generate input for CE assessments. If neither observed data nor modelling results are available as inputs, assessments often rely on expert knowledge, where the quantification of uncertainties is even more difficult, and therefore often ignored, aggravating the deep uncertainties that are present in the assessment.

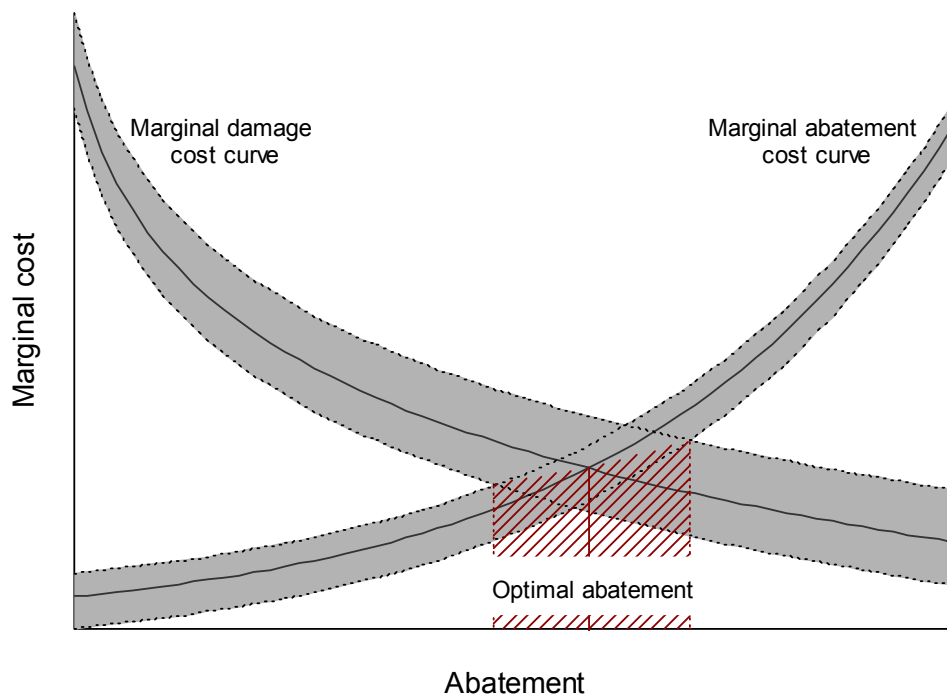
*Table 13. An inventory of uncertainties in the economic assessment of agricultural GHG mitigation*

Model inputs	Source of uncertainty	System	Type of uncertainty
Global warming potential (GWP) of GHGs	Variability of the atmospheric processes	Biogeochemical	Statistical
	Modelling future atmospheric processes	Biogeochemical	Statistical and deep
	Choice of GWP metric	Economic	Value
Agricultural activity levels (e.g. 0.9 M ha)	Historic agricultural activity, prices and other economic variables	Economic	Statistical



Model inputs	Source of uncertainty	System	Type of uncertainty
permanent grassland)	Modelling future changes in farming activities as a function of demographic and economic changes	Economic and political	Statistical and deep
GHG abatement achievable by MMs (e.g. 0.1 t CO <sub>2</sub> e/ha/year) AND Biophysical interactions between MMs (e.g. 10% reduction in GHG abatement of MM A if applied together with MM B)	Variability in weather and in the soil processes involved in N <sub>2</sub> O emissions	Biogeochemical	Statistical and deep
	Modelling future soil processes	Biogeochemical	Statistical and deep
	Modelling how farmers implement the MMs in practice	Behavioural	Statistical and deep
	Modelling future changes in abatement efficacy of MMs	Technological	Statistical and deep
Applicability of MMs (e.g. % of land area)	Weather and soil types	Biogeochemical	Statistical
	Current and future type of farming systems (e.g. organic)	Economic	Statistical and deep
Likely additional uptake of MMs by farmers (e.g. 45% of land area)	Current farm management practices	Economic	Statistical
	Variability in farmers' behaviour	Behavioural	Statistical
	Modelling farmers' future behaviour	Behavioural	Statistical and deep
	Modelling future changes in the economy and farming	Economic and political	Statistical and deep
Annualised net cost of MMs (e.g. £1.40 /ha/year)	Historic prices and other economic variables	Economic	Statistical
	Modelling future changes in prices and farming practices	Economic, technological	Statistical and deep
	Modelling future farm finances	Economic	Statistical and deep
	Choice of discount rate	Economic	Value
Carbon price threshold (£29 / t CO <sub>2</sub> e)	Modelling future atmospheric processes	Biogeochemical	Statistical and deep
	Modelling the physical impacts of climate change	Biogeochemical and economic	Statistical and deep
	Modelling future changes in the economy	Economic and political	Statistical and deep
	Choice of discount rate	Economic	Value

A quantitative assessment of the statistical uncertainty of the CE of MMs is presented in the following two sections via a case study of the Scottish agricultural MACC. MACCs represent the marginal cost of emission reduction (i.e. the cost of each additional unit of GHG abatement). The economically optimal abatement level is determined by the interception point of the MACC and the marginal damage cost curve, which measures the marginal cost of GHG emissions to the society (i.e. the social cost of an additional unit of GHG in the atmosphere). Uncertainty in the MACC and in the marginal damage cost curve result in uncertainty in the economic optimum (Figure 13). A MACC that is comprised of alternative technologies as MMs (engineering MACC) is likely to have additional uncertainties in the abatement potential, the cost of each MM and in the ranking of the MMs. Thus the uncertainty information becomes highly relevant if a MACC is used in the policy process, for example to inform decisions on prioritising certain MMs for policy support.



*Figure 13. Effect of uncertainty on the optimal abatement level*

*Based on Smith and Stern (2011). Grey areas delimited by dashed black lines show the confidence interval of the marginal damage and marginal abatement cost curves with solid black lines representing mean values. Red area delimited by dashed red lines show the confidence interval of the economically optimal abatement with solid red line representing mean value.*

## 5.5 Data and Methods

Moran *et al.* (2011b) estimated the annual net costs and annual GHG abatement potential of MMs applicable in the UK agriculture, and calculated the ratio of these to obtain the annual CE of the MMs (£ t CO<sub>2</sub>e<sup>-1</sup>). The additional annual GHG abatement above the abatement forecast in the BAU scenario was calculated for the years 2012, 2017 and 2022, considering interactions between the MMs (i.e. possible synergies and trade-offs in mitigation if more than one MM is implemented at the same time on the same farm). Future predictions included changes in agricultural activities and prices, but not changes in the climate. Four measure uptake scenarios were modelled, reflecting different assumptions about the future policy environment: low, central and high feasible potential plus a maximum technical potential, assuming uptake levels of 7-18%, 45%, 85-92% and 100% respectively – see Moran *et al.* (2008) for a detailed description. In this paper we focus on the GHG MMs in Table 14 (for a description see Moran *et al.* (2008)):

*Table 14. Mitigation measures analysed in this Chapter*

No.	Mitigation measure
MM1	Using biological fixation to provide nitrogen inputs
MM2	Reducing nitrogen fertiliser
MM3	Improving land drainage
MM4	Avoiding nitrogen application in excess
MM5	Using manure nitrogen to its full extent
MM6	Introducing of new species (including legumes)
MM7	Improving the timing of mineral nitrogen application
MM8	Improving the timing of slurry and poultry manure application
MM9	Using controlled release fertilisers
MM10	Using nitrification inhibitors
MM11	Adopting systems less reliant on inputs
MM12	Adopting plant varieties with improved N-use efficiency
MM13	
MM14	Using reduced tillage and no-till techniques
MM15	Using composts, straw-based manures in preference to slurry

As information on the statistical uncertainty of the input variables needed to build a MACC was not available we conducted an uncertainty assessment rather than an uncertainty analysis. In other words, we assessed the impact of uncertainty on the results, rather than trying to quantify the level of uncertainty within the results. An uncertainty analysis would

rely on existing quantitative information regarding the uncertainty of the MACC input variables, commonly in the form of probability density functions.

Given the lack of quantitative information regarding the level of input uncertainty, three level of uncertainty were assigned for each input variable ( “wide”, “medium” and “narrow” PDF), which are respectively based on assuming that levels of uncertainty are high, medium or low. Importantly, the uncertainty of the carbon price threshold was not considered in the analysis. The three confidence intervals assigned to of each of the input variables were based on the authors’ judgment. Further, three different parametric distributions were considered in each case: the censored normal, truncated normal and triangular distributions. These distributions were considered to investigate the effect of the shape of the PDF; they all allow the boundaries of the parameter space to be dealt with in a particular way (i.e. the fact that uptake must lie between 0 and 1). The three parametric distributions each describe the PDF in terms of two parameters - the mode (the value associated with the highest probability) and the confidence interval (CI) (the range that includes 95% of probability, or, for the triangular distribution, 100% of probability). The mode is taken to be the value of the each input variable that was originally used in the MACC, and the CI is specified separately for each input variable and level of uncertainty (Table 15). For some input variables (e.g. net cost) the CI is assumed to be a multiple of the mode, whilst for others (e.g. uptake) it is assumed to have a value that is unrelated to the mode.

The three parametric distributions differ in terms of their shape. For the triangular distribution, probability is a linear function of distance from the mode. For the censored normal and truncated normal distribution, it is assumed that the distribution of probabilities can be represented by a normal distribution bounded by the parameter space of the input variable. These two distributions differ solely in whether there is a non-zero probability of obtaining values that lie exactly at the boundaries of the parameter space; the censored normal allows this, the truncated normal does not. The two distributions are equivalent to each other – and equivalent to a conventional normal distribution – for those input variables that have no boundaries on their parameter space (e.g. net cost, and, for some MMs, abatement rate).

Table 15. Characteristics of the three levels of uncertainty assigned to the input variables of the MACC model

Uncertainty source	Description and unit	Parameter space of the input variable	Confidence interval <sup>1</sup>		
			Wide PDF	Medium PDF	Narrow PDF
N <sub>2</sub> O GWP	100 year GWP [kg CO <sub>2</sub> e (kg N <sub>2</sub> O) <sup>-1</sup> ]	(0, ∞)	Mode * 0.6	Mode * 0.4	Mode * 0.2
Activity levels	Areas of land under different type of crops (four crop categories) [ha]	(0, ∞)	Mode * 0.6	Mode * 0.4	Mode * 0.2
Applicability	Biophysical feasibility of applying an MM on a land category [-]	(0, 1)	1.0	0.6	0.2
Uptake	Level of implementation of a MM by farmers across Scotland, on land areas where the MM is applicable [-]	(0, 1)	1.0	0.6	0.2
Interaction factors	Factor assigned to each possible pairs of MMs, describing the synergies and trade-offs in the GHG effectiveness of the MMs [-]	(0, ∞)	1.0	0.6	0.2
Abatement rate	Technical GHG effectiveness of the MMs [t CO <sub>2</sub> e ha <sup>-1</sup> year <sup>-1</sup> ]	(0, ∞) (-∞, ∞)	Mode * 4	Mode * 2	Mode
Net cost	Difference between the gross margin of the farm with and without the MM applied, calculated with a profit maximising farm model [£ ha <sup>-1</sup> year <sup>-1</sup> ]	(-∞, ∞)	Mode * 4	Mode * 2	Mode

<sup>1</sup> The CI is 95% for the censored normal and the truncated normal distribution, 100% for the triangular distribution

Activity levels and the global warming potential of N<sub>2</sub>O were assumed to have the lowest uncertainty;– the former based on the fact that annual farming statistics in Scotland are estimated with relatively high certainty, and the latter based on the confidence range of GWPs reported by the IPCC (IPCC 2007). Measure applicability, uptake and interaction factor (IF) values were based on expert judgement in the original exercise (Moran *et al.* 2008), therefore greater levels of uncertainty were assigned to them than to GWP and activity levels. Applicability and uptake can be of any value between 0 and 1, where 1 represents applicability on 100% of agricultural land, and 100% uptake, respectively. Most of the IFs fall between 0 and 1. The IF values that represent synergies, such as for the interaction effect between ‘Improving land drainage’ and ‘Using nitrification inhibitors’, have values above 1. As the uncertainty of applicability, uptake and interaction factors is assumed not to be proportional to their value, their uncertainty was expressed in absolute terms. Net costs of MMs, derived from a farm level financial model with no information on

their uncertainty, were assigned relatively high levels of uncertainty. Abatement rates, which were based on expert judgement, were similarly assigned high levels of uncertainty. Among the abatement rates of the 15 MMs, seven were assumed to be non-negative. The remaining eight MMs were assumed to have some probability for negative values. That is, with a certain probability these MMs were assumed to increase, rather than decrease, GHG emissions<sup>2</sup>.

Statistical uncertainty of the input variables was propagated through the model via Monte Carlo analysis. The Monte Carlo analysis for each combination of year, uptake scenario, level of uncertainty, parametric distribution and uncertainty source involved simulating 1,000 sets of input variable values using the relevant PDFs, and then using each set of simulated input variables for calculation of the MACC in order to generate a PDF for the MACC outputs. The key outputs collected were the distribution of the ranking of each MM (in terms of their annual CE), and the distribution of the economically optimal abatement potential. The optimal abatement potential corresponds to the aggregated annual abatement potential of all of the GHG MMs of the MACC with a CE value below the marginal damage cost curve. To represent the marginal benefit of mitigation, a constant value for damage cost was applied using the shadow price of carbon (SPC) with a value of £29 (CO<sub>2</sub>e t)<sup>-1</sup> (Price *et al.* 2007). Monte Carlo simulations were run for all 3 x 4 x 3 x 3 x 8 = 864 combinations of

- year: 2012, 2017, 2022;
- uptake scenario: low feasible potential (LFP), central feasible potential (CFP), high feasible potential (HFP) and MTP (maximum technical potential);
- level of uncertainty: narrow PDF, medium PDF, wide PDF;
- parametric distribution: censored normal, truncated normal, triangular; and
- uncertainty source: N<sub>2</sub>O GWP, activity level, applicability, uptake, interaction factors, abatement rate, net cost, or all seven sources combined.

Lacking any quantitative information on possible dependence between the different sources of uncertainty, it is assumed that the uncertainties associated with the different sources are independent for the simulations that combine all seven sources.

---

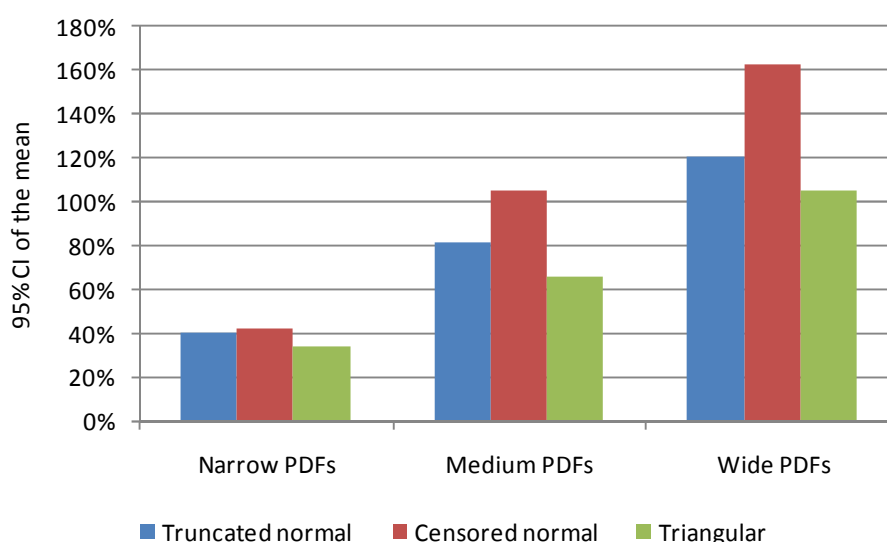
<sup>2</sup>The eight mitigation options with non-zero probabilities for negative values: ‘Improving land drainage’, ‘Introduction new species (including legumes)’, ‘Improving the timing of mineral nitrogen application’, ‘Improving the timing of slurry and poultry manure application’, ‘Adopting plant varieties with improved N-use efficiency’, ‘Separating slurry applications from fertiliser applications by several days’, ‘Using reduced tillage and no-till techniques’, ‘Using composts, straw-based manures in preference to slurry’.

## 5.6 Results

### 5.6.1 Uncertainty of the economically optimal GHG abatement

The level of economically optimal GHG abatement is a key result from MACCs. We quantify uncertainty in the economically optimal GHG abatement by taking the 95% confidence interval of the mean and expressing it as a proportion of the mean; while keeping the carbon price threshold constant. For example, for the scenario 2022, central feasible potential, censored normal distribution, medium PDF and all uncertainty sources combined the mean of the economically optimal GHG abatement is 850 kt CO<sub>2</sub>e and the standard deviation is 270 kt CO<sub>2</sub>e (32% of the mean). The 95% confidence interval of the mean is therefore  $\pm 32\%$  (normality can be assumed as the skewness of the distributions is between  $-0.5$  and  $0.5$ ).

As an example, Figure 14 shows the uncertainty associated with different levels of uncertainty and parametric distributions for the simulation that combine all seven uncertainty sources, the CFP uptake scenario and year 2022. The uncertainty in the wide PDF is, unsurprisingly, higher than that in the medium PDF, and this is, in turn, higher than that in the narrow PDF. The censored normal distribution produces higher estimates of uncertainty than the other two parametric distributions because it is the only model to allow a non-zero probability where the true value will be equal to the boundary of the parameter space of the input variable (e.g. 0 or 1, for uptake). The estimated uncertainty for the truncated normal and triangular distributions is similar, but uncertainty is generally lowest for the triangular distribution.



*Figure 14. The 95% CI of the mean of the economically optimal GHG abatement for the three parametric distributions: truncated normal, censored normal and triangular, for the three levels of uncertainty (narrow, medium and wide PDFs). CFP uptake scenario, in year 2022, for all uncertainty sources combined*

When propagating the uncertainties of all the input variables across all combinations of year, uptake scenario, level of uncertainty and parametric distribution, the 95% CI was between 10-107% of the mean of the economically optimal GHG abatement (Table 16). The lowest uncertainty exists for the maximum technical potential in 2022 with narrow PDFs and triangular distribution, and the highest uncertainty is found for the low feasible potential in 2012, with wide PDFs and censored normal distribution. In general, the uncertainty of this output metric decreases with the increasing level of uptake as we move from uptake scenario LFP to uptake scenario MTP, and also as the results are projected further into the future. The change across the years originates from a decline in the output uncertainty caused by the uncertainty in the input variables, namely abatement rate, applicability, interaction factors and, most of all, the level of uptake. The diminishing influence of uptake uncertainty on the output uncertainty is partly due to the assumption of linearly increasing uptake through time, which has a constant uncertainty associated with it. The change across the uptake scenarios is a similar phenomenon, as the uptake scenarios assume an increasing uptake rate from the low feasible towards the maximum technical potential.



*Table 16. Lowest and highest value of the 95% CI of the mean of the economically optimal GHG abatement for different simulations*

Uncertainty source	Parametric distribution	Level of uncertainty	Year	Uptake scenario	95% CI of the mean	
					Lowest value	Highest value
All sources combined	any	any	any	any	0.096	1.073
Applicability	any	any	any	any	0.034	0.217
Abatement rate	any	any	any	any	0.067	0.332
Net costs	any	any	any	any	0.007	0.100
Uptake	any	any	any	any	0.010	0.671
Interaction factors	any	any	any	any	0.024	0.234
GWP	any	any	any	any	0.039	0.171
Activity level	any	any	any	any	0.029	0.116
all	Censored normal	any	any	any	0.119	1.073
all	Truncated normal	any	any	any	0.122	0.699
all	Triangular	any	any	any	0.096	0.575
all	any	Narrow PDFs	any	any	0.096	0.514
all	any	Medium PDFs	any	any	0.221	0.769
all	any	Wide PDFs	any	any	0.400	1.073
all	any	any	2012	any	0.126	1.073
all	any	any	2017	any	0.109	0.871
all	any	any	2022	any	0.096	0.811
all	any	any	any	LFP	0.168	1.073
all	any	any	any	CFP	0.106	0.941
all	any	any	any	HFP	0.100	0.750
all	any	any	any	MTP	0.096	0.787

The contribution of the uncertainty in each uncertainty source to that of the economically optimal abatement was examined by propagating the uncertainty of one source at a time, for all the three years, four uptake scenarios and three levels of uncertainty – see Table 16, and examples on Figure 15 (2022, CFP). The uncertainties in the abatement rate and in the uptake were the most important contributors in the majority of simulations (the 95% CI ranged between 7-33% and 1-67% of the mean for these two sources of uncertainty, respectively). In simulations with a low level of uptake (year 2012 or uptake scenario LFP) the uncertainty associated with the uptake of MMs was more significant whereas in simulations with a high level of uptake the uncertainty in the abatement rate caused higher uncertainty in the output. The uncertainties in the net cost and activity level were usually the least important in the output uncertainty (the 95% CI was between 0.7-10% and 3-12% of the mean for these two UC sources, respectively).

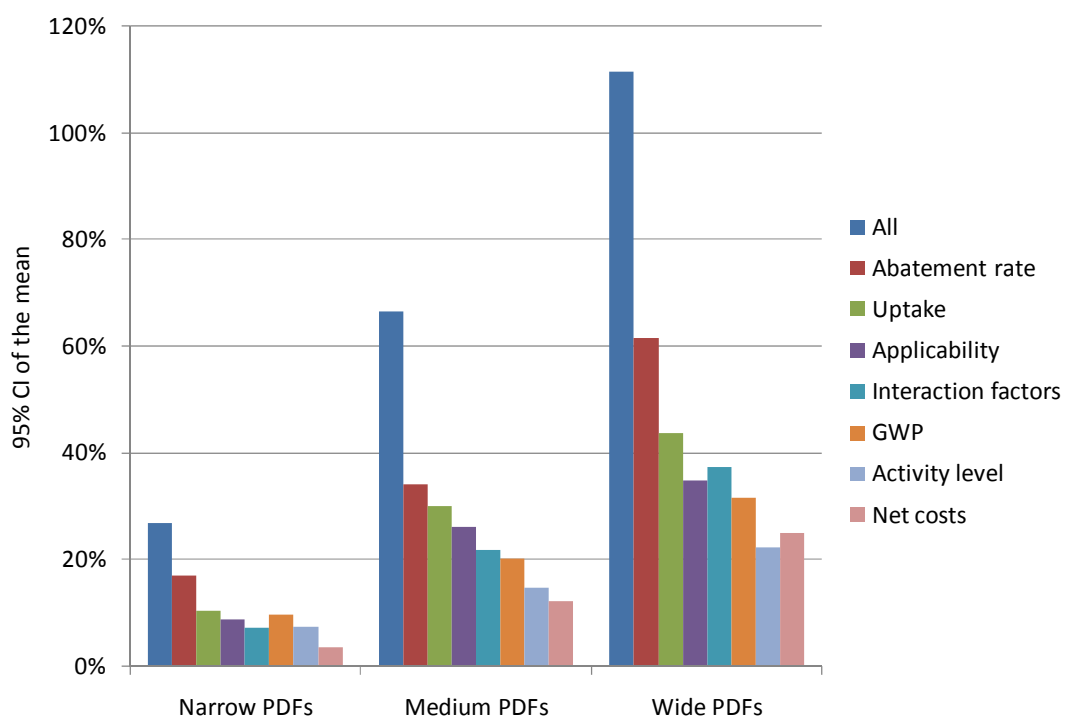


Figure 15. The 95% CI of the mean of the economically optimal GHG abatement by propagation of uncertainty of individual uncertainty sources and all uncertainty sources combined, for three levels of uncertainty

Scenario: truncated normal distribution, 2022, CFP uptake

## 5.6.2 Uncertainty in the ranking of the mitigation measures

The input uncertainty results in uncertainty in the ranking of MMs due to the uncertainty in their CE and in the interaction factors. Figure 16 reveals that this uncertainty can be relatively high in the ranking of some MMs, especially if wide PDFs are propagated through the model. For example the ranking of ‘Improving land drainage’ has a wide, trimodal distribution. Although it is most likely to be ranked as the third best MM, it still has an 8% probability that its CE is higher than the shadow price of carbon (depicted by the area under the curve to the right of the SPC). MMs with CE closest to 0 are the least uncertain in terms of ranking, which can be partly explained by the PDFs assigned to the net cost and the abatement rate, both of which are proportional to the mode. Despite the uncertainty in the individual ranking of the MMs, the set of MMs that are estimated to be cost-effective is relatively stable.

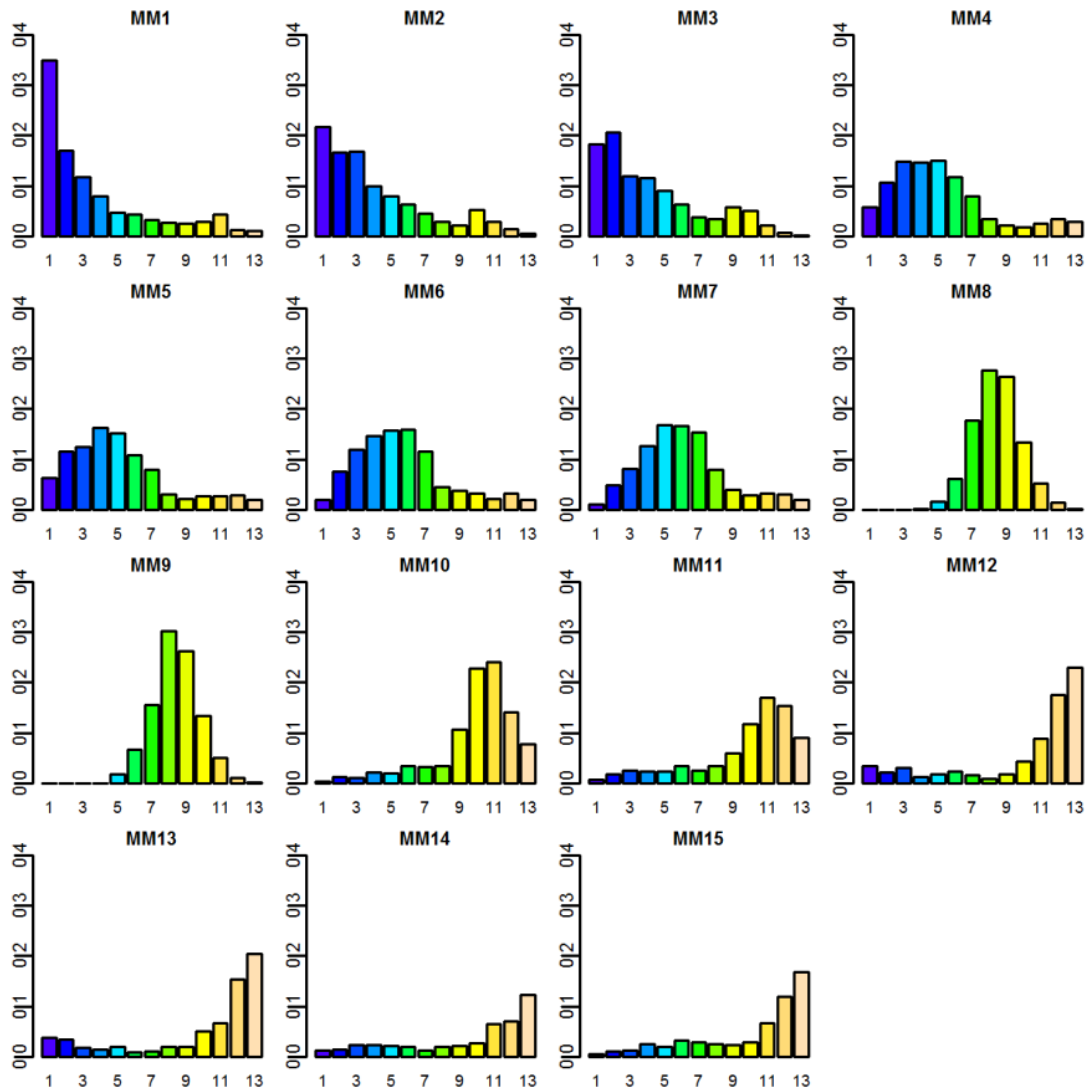


Figure 16 The probability of the ranking of each MM

Scenario: truncated normal distribution, all uncertainty sources combined, 2022, CFP uptake, wide PDFs. X axis: ranking (rank 14 and 15 are not shown; e.g. rank 1 means that the MM was the first on the left on the MACC), Y axis: probability. MM1: Improving the timing of mineral nitrogen application, MM2: Adopting plant varieties with improved N-use efficiency, MM3: Improving land drainage, MM4: Using reduced tillage and no-till techniques, MM5: Improving the timing of slurry and poultry manure application, MM6: Avoiding nitrogen application in excess, MM7: Using manure nitrogen to its full extent, MM8: Separating slurry applications from fertiliser applications by several days, MM9: Using composts, straw-based manures in preference to slurry, MM10: Using nitrification inhibitors, MM11: Introduction of new species (including legumes), MM12: Using controlled release fertilisers, MM13: Reducing N nitrogen fertiliser, MM14: Using biological fixation to provide nitrogen inputs, MM15: Adopting systems less reliant on inputs

## 5.7 Discussion and conclusion

The qualitative assessment of the MACC revealed the numerous sources of uncertainty in the economic assessment of agricultural GHG mitigation. These complex assessment exercises incorporate many aspects of uncertainty - from modelling the biophysical processes through to economic, political and behavioural aspects. Deep uncertainties are present in relation to every input variable, and available information on the statistical uncertainties is often limited. Part of the underlying data is easily accessible (such as basic statistics on current activity levels), at least in countries where agricultural statistical data are commonly collected. But even for such data, uncertainty information is not commonly reported. There is an extensive literature on technical abatement potential of agricultural GHG MMs, enabling the use of meta-analysis to quantify statistical uncertainty. But reporting practices regarding uncertainty tend not to be rigorous and consistent (Buckingham *et al.* 2014). To improve these practices, guidance could be developed on how the statistical uncertainty of experimental and modelling results can be reported to better serve economic assessments. Furthermore, there is a need to decompose quantitatively the abatement potential of the MMs as provided by the various biophysical processes behind the GHG mitigation effect. Ideally, this decomposition would allow expressing the MMs' abatement potential according to the IPCC emission calculation methodologies.

As an *ex ante* assessment, MACCs often draw on inputs derived via modelling exercises such as soil or farm models. The analysis and reporting of uncertainty associated with such model outputs is typically ignored. Due to this lack of information an uncertainty analysis was not possible. However, useful recommendations can still be drawn from an uncertainty assessment, whereby possible uncertainty in the assessment results is provided and areas for further improvement are identified.

The quantitative assessment of the MACC shows that the uncertainty in the economically optimal abatement becomes high if high and medium levels of uncertainty are assumed: the 95% CI was between  $\pm 40$ -107%,  $\pm 22$ -77% of the mean, respectively, across the years and uptake scenarios and parametric distributions. Assuming low uncertainty in the input variables results in much lower uncertainty of this output metric; the 95% CI was between  $\pm 10$ -51% of the mean across the years and uptake scenarios and parametric distributions. However, the ranking of the measures is relatively robust, especially for MMs with CE estimated to be below a carbon price of £29 (CO<sub>2</sub>e t)<sup>-1</sup>. Although there is a high level of uncertainty regarding estimates of economically optimal abatement potential, these findings

imply that the choice of MMs to be implemented on farms – from a CE perspective –, can still be robust. This finding corresponds to Gibbons *et al.* (2006), who found that the total emissions from farms are very uncertain, but that the relative effects of MMs, expressed as a proportion of total farm emissions, had a lower degree of uncertainty.

In terms of the contribution of uncertainties in the input variables to the uncertainty in economically optimal abatement potential, abatement rate and uptake are the most important input variables. These two input variables, along with applicability, net cost and interaction factors, have the largest degree of input uncertainty. Input variables that have both high levels of uncertainty and a large contribution highly to the uncertainty in the outputs are key to reducing uncertainty in the outputs (Heijungs 1996). Following this, Figure 17 categorises all input variables in terms of their potential role in reducing uncertainty in the analysis, based on their contribution to the uncertainty in the optimal abatement potential and the estimated effort needed to reduce the input uncertainty.

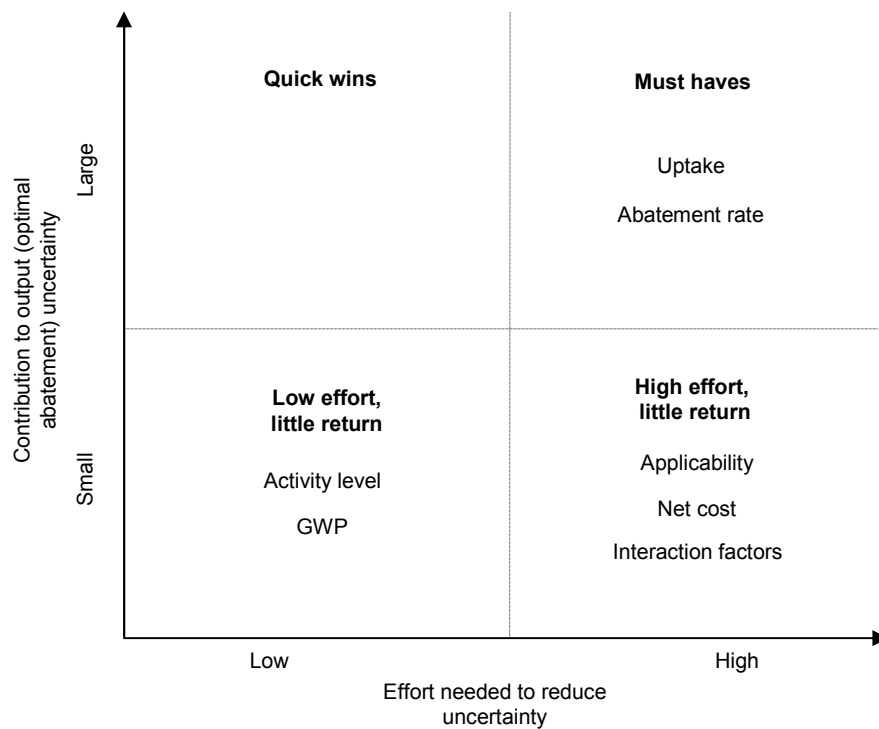


Figure 17. Uncertainty assessment of the input variables

In considering opportunities to reduce the uncertainty in analyses of agricultural GHG mitigation, the effort associated with reducing uncertainty must be weighed against the benefits of more robust predictions. The data gaps regarding uncertainties related to inputs for MACC analysis are very large; considering both biophysical and socio-economic variables. Improving scientific reporting practice to include quantified data about the statistical uncertainty in underlying research is likely to be one of the most efficient ways to address this issue. Estimates on the statistical uncertainty of GWP and activity level are available from literature on climate research and from the agricultural statistic offices. Including this type of information in the uncertainty assessments would require low effort, though it is likely to yield minor improvements in the outcome uncertainty. In contrast, uncertainty regarding the level of uptake, the abatement rate, the applicability and net cost of MMs along with the interactions between them requires further research, with the first two inputs having the highest potential to reduce the uncertainty of the optimal abatement potential. On-going research on biophysical processes of GHG mitigation is starting to place greater emphasis on providing more information on the abatement rate and, to a limited degree, on the biophysical interactions between MMs. Such information will be most useful if it is accompanied by estimates of uncertainty. Similar improvements are needed in economic analyses in order to reduce uncertainty associated with cost estimates, and in order to improve the robustness of predictions of future changes in agriculture and land use. Uncertainty in the level of uptake can be improved through a better understanding of behavioural processes and of the effects of PIs on farmers' choices. Applicability values are ultimately based on the opinions of agronomic experts. Formal elicitation of uncertainty in this case is also possible, although resource intensive. Overall, it is likely that the uncertainties in biophysical and economic modelling will become more explicit in the future, reducing the extent of uncertainty in integrated modelling.

Like other integrated assessment tools, MACCs accumulate uncertainties. Input data might include agricultural statistics, meta-analysis of field experiments, results from biophysical and financial models, results from expert elicitation exercises, or assumptions based on the judgment of researchers. These inputs all have their underlying uncertainties, which are only partially quantifiable. However, assessing the importance of these uncertainties and the extent to which they can be reduced is an important step in the development of more robust policies.

As a general guideline, the analysis in this paper concurs with observations previously made on the need to improve the reliability of integrated assessments in relation to policy (Kann and Weyant 2000, Smith and Stern 2011); specifically the provision of

- Probability-weighted values ('implied probabilities') of the outputs;
- information on where the model results provide reliable information (i.e. what are the boundaries of the model's relevance);
- key inputs driving the uncertainty of the outputs;
- the extent of unquantifiable uncertainty.

To address high uncertainty in the economically optimal GHG abatement Scottish soils, this paper recommends a research focus on the potential mitigation effects of the MMs and at the likelihood of farmers' future uptake. In terms of supporting particular GHG MMs, it is encouraging that the ranking of the MMs is robust, as is their CE relative to the shadow price of carbon, therefore decisions on the prioritisation of MMs in policy have a robust basis.

## 6 Paper V. Linking an economic and a life-cycle analysis biophysical model to support agricultural greenhouse gas mitigation policy

Vera Eory, Michael MacLeod, Shailesh Shrestha and David Roberts

Research Division, SRUC, West Mains Road, Edinburgh, EH9 3JG, UK

This piece of research was published in the German Journal of Agricultural Economics in 2014, vol 63, pp. 133-142. My contribution to the research was the design of the methodology and major input to the analysis and discussion of results.

### 6.1 Abstract

Greenhouse gas (GHG) mitigation is one of the main challenges facing agriculture, exacerbated by the increasing demand for food, in particular for livestock products. Production expansion needs to be accompanied by reductions in the GHG emission intensity (EI) of agricultural products, if significant increases in emissions are to be avoided. Suggested farm management changes often have systemic effects on farm, therefore their investigation requires a whole farm approach. At the same time, changes in GHG emissions arising off-farm in food supply chains (pre-, or post-farm) can also occur as a consequence of these management changes. A modelling framework that quantifies the whole-farm, life-cycle effects of GHG MMs on emissions and farm finances has been developed. It is demonstrated via a case study of sexed semen on Scottish dairy farms. The results show that using sexed semen on dairy farms might be a cost-effective way to reduce emissions from cattle production by increasing the amount of lower EI ‘dairy beef’ produced. It is concluded that a modelling framework combining a GHG life cycle analysis model and an economic model is a useful tool to help designing targeted agro-environmental policies at regional and national levels. It has the flexibility to model a wide variety of farm types, locations and management changes, and the LCA-approach adopted helps to ensure that GHG emission leakage does not occur in the supply chain.



## 6.2 Introduction

Reducing greenhouse gas emissions arising from agricultural activities remains a challenge as the world is starting to experience the consequences of a changing climate (IPCC 2013b, Renwick *et al.* 2013) and at the same time food production is facing major challenges both in demand for land-based products and also in terms of production constraints (Foresight 2011). Satisfying growing demand for livestock products will lead to significant increases in the greenhouse gas emissions from the sector unless the EI (i.e. the GHG emissions arising from the production of a unit of output, e.g. kg CO<sub>2</sub>e (litres of milk)<sup>-1</sup>) can be reduced.

Globally, cattle milk is the largest source of livestock protein and global milk demand is forecast to increase by 80% by 2050, relative to 2005/7 demand (Alexandratos and Bruinsma 2012). The greenhouse gas emissions arising from global milk production were quantified by (Gerber *et al.* 2010) and increasing attention is being paid to finding ways of reducing the EI of milk production.

Numerous management changes and technologies have been proposed to reduce on-farm emissions from livestock (see for example (Bellarby *et al.* 2013, Cottle *et al.* 2011, Hristov *et al.* 2013)). A few measures only affect one emission source on the farm; for example reducing excess nitrogen fertiliser decreases N<sub>2</sub>O emission without any further implications on the other activities on farm. However, many measures can have system-wide effects, e.g. changing the ration can lead to changes in enteric CH<sub>4</sub> emissions, changes in volatile solid and N excretion rates (with consequent impacts on manure CH<sub>4</sub> and N<sub>2</sub>O emissions), and also changes in the amount of meat or milk produced. The use of whole farm modelling approaches provides a powerful tool for analysing the system-wide effects of GHG MMs on emissions and farm financial performance.

In addition to the systemic effects within the farm outlined above, interactions can also occur along the supply chain. For example, changing the way in which inputs such as synthetic fertilisers and feed materials are produced can change the emission intensities of the final commodities produced. These effects can be captured by using a life cycle analysis approach in the evaluation of MMs.

Various whole farm models and modelling frameworks have been developed, mostly for one or two particular farming systems (see reviews of the ruminant systems by (Crosson *et al.* 2011, Schils *et al.* 2007)), while some are capable of simulating different farming systems

(Louhichi *et al.* 2010, Neufeldt and Schafer 2008). However, LCA GHG calculations are rarely provided by these tools, therefore in this paper we outline an approach which is capable of simulating management changes on various farm systems to provide ex-ante evaluation of LCA GHG emissions and economic effects.

The farm level modelling framework presented here consists of the Global Livestock Environmental Assessment Model life-cycle GHG emission model (MacLeod *et al.* 2013) and ScotFarm, an optimising farm level model based on a linear programming farm economic model described by Shrestha (2004). Within this framework, the emissions, production and farm income can be calculated with and without MMs, thus enabling the CE of measures and the interactions between the measures to be quantified for specific-farm systems and locations. This paper provides an explanation of the approach and a case study of sexed semen on Scottish dairy farms. Finally, the strengths and weaknesses of the approach and MMs for future development are discussed.

## 6.3 Methodology

### 6.3.1 Global Livestock Environmental Assessment Model

The Global Livestock Environmental Assessment Model (GLEAM) is an LCA model developed by the UN Food and Agriculture Organisation (MacLeod *et al.* 2013). It simulates processes within livestock production systems in order to assess their environmental performance. The current version of the model (V1.0) focuses primarily on the quantification of GHG emissions and includes: (a) pre-farm emissions arising from the manufacture of inputs; (b) on-farm emissions during crop and animal production; and (c) post-farm emissions arising from the processing and transportation of products to the retail point. Emissions and food losses that arise after delivery to the retail point are not included. While gases of minor importance have been omitted, the three major GHG in agriculture are included, namely: (1) CH<sub>4</sub> (mainly from enteric fermentation, manure storage and rice cultivation), (2) N<sub>2</sub>O (from soils and manure storage) and (3) CO<sub>2</sub> from (a) the combustion of fossil fuels on-farm (e.g. in tractors and generators) and off-farm (in the manufacture of inputs, including mineral fertilisers, and in post-farm processing and transport) and (b) land use change. CO<sub>2</sub> from the short biological cycles such as respiration and aerobic decomposition are not included. GLEAM calculates:

- total production of the main livestock commodities, i.e. meat, milk and eggs

- the total greenhouse gas emissions arising from that production
- the EI of each commodity.

A brief overview of the model elements is given below, and values for selected parameters given in Table 17.

The herd module starts with the total number of animals of a given species and system. It determines the herd structure (i.e. the number of animals in each cohort, and the rate at which animals move between cohorts) and the characteristics of the average animal in each cohort (e.g. weight and growth rate). The manure module calculates the rate at which total excreted N is applied to crops, accounting for losses during storage. The feed module calculates key feed parameters, i.e. the nutritional content and emissions per kg of the feed ration. The system module calculates each animal cohort's energy requirement, and the total amount of meat, milk and eggs produced each year. It also calculates the total annual emissions arising from manure management, enteric fermentation and feed production. The allocation module combines the emissions from the system module with the emissions calculated outside GLEAM, i.e. emissions arising from (a) direct on-farm energy use; (b) the construction of farm buildings and manufacture of equipment; and (c) post-farm transport and processing. The total emissions are then allocated to the co-products (e.g. meat and milk) and the EI of the commodities are calculated.

*Table 17. Value of selected parameters for lactating cows*

Category	Parameter	Value	Notes <sup>1</sup>
Ration	Ration digestibility (%)	78%	Based on a ration of 62% fresh grass, 38% compound feed
Ration	Ration emissions intensity (kg CO <sub>2</sub> e (kg DM) <sup>-1</sup> )	1.4	IPCC (2006) Tier 1
Intake	NE requirement (MJ cow <sup>-1</sup> day <sup>-1</sup> )	121.8	IPCC (2006) Tier 2
Intake	Feed intake (kg DM cow <sup>-1</sup> day <sup>-1</sup> )	15.4	IPCC (2006) Tier 2
Output	VS excretion (kg cow <sup>-1</sup> day <sup>-1</sup> )	3.64	IPCC (2006) Tier 2
Output	N excretion (kg N cow <sup>-1</sup> day <sup>-1</sup> )	0.39	IPCC (2006) Tier 2
Output	Enteric CH <sub>4</sub> (kg CH <sub>4</sub> cow <sup>-1</sup> year <sup>-1</sup> )	109	IPCC (2006) Tier 2
Manure	CH <sub>4</sub> conversion factor (%VS)	6.3%	IPCC (2006) Tier 2, based on 68% PRP, 32% slurry (no cover)
Manure	Manure CH <sub>4</sub> (kg CH <sub>4</sub> cow <sup>-1</sup> year <sup>-1</sup> )	13.4	IPCC (2006) Tier 2
Other	Average annual temperature (°C)	10	Assumption
Other	CH <sub>4</sub> conversion factor (Ym) (%)	6.5%	IPCC (2006, Table 10.12)
Other	B <sub>0</sub> (m <sup>3</sup> CH <sub>4</sub> (kg VS) <sup>-1</sup> )	0.24	IPCC (2006, Table 10.A4)

<sup>1</sup> IPCC (2006) parameters are based on (IPCC 2006)

### 6.3.2 ScotFarm

ScotFarm, a profit optimising financial model developed at SRUC, is based on a farm level dynamic linear programming model which is described in detail in Shrestha (2004).

Modified versions of farm level linear programming models have been used in a number of farm level analyses of Irish agriculture (Hennessy *et al.* 2008, Shrestha *et al.* 2013, Shrestha and Hennessy 2006, Shrestha *et al.* 2007). ScotFarm assumes that all farmers are profit oriented and maximise farm net income within a set of limiting farm resources. It consists of four production systems; dairy, beef, sheep and arable. These systems are constrained by the land, labour, feed and stock replacement available to a farm. The total land available to a farm is fixed. Farms are allowed to buy in feeds, animal replacements and hire labour if required. The farm net income is comprised of the accumulated revenues collected from the final product of the farm activities (crops, animals and milk) plus farm payments minus costs incurred for inputs under those activities. The input costs are replacement costs for livestock, variable costs including labour, feed and veterinary costs and overhead costs on farms.

The model consists of all the major crops in Scotland. The initial land under these crops in each farm is based on farm level data of the 2010/11 Farm Account Survey of Scotland (see section 6.3.4); however, the model allows land to be reallocated between these crops as well as transferred to grass production. The stocking rate on each farm is also fixed to the farm level data assuming that all farms were operating under optimum stocking rate. The dairy system has a four year replacement structure where dairy animals are culled after every four years. Similarly beef and sheep systems follow a two year replacement structure. The animals are replaced by on-farm or off-farm replacement stocks. A feed module, based on (Alderman and Cottrill 1993) is used in the model to determine feed requirements for each of the animals on a farm based on type, age and production level of the animal. Feeds available to the livestock on farm are fresh grass, grass silage, grass hay, maize silage, grains, straw, beet and concentrate feeds.

### 6.3.3 Harmonising GLEAM and ScotFarm

Model parameters, input variables and modules are harmonised in GLEAM and ScotFarm in order to simulate the model farms and the MM's effect in parallel in both models. The herd structure, land use and feed ration composition are optimised in ScotFarm, and then exported to GLEAM (see Figure 18).

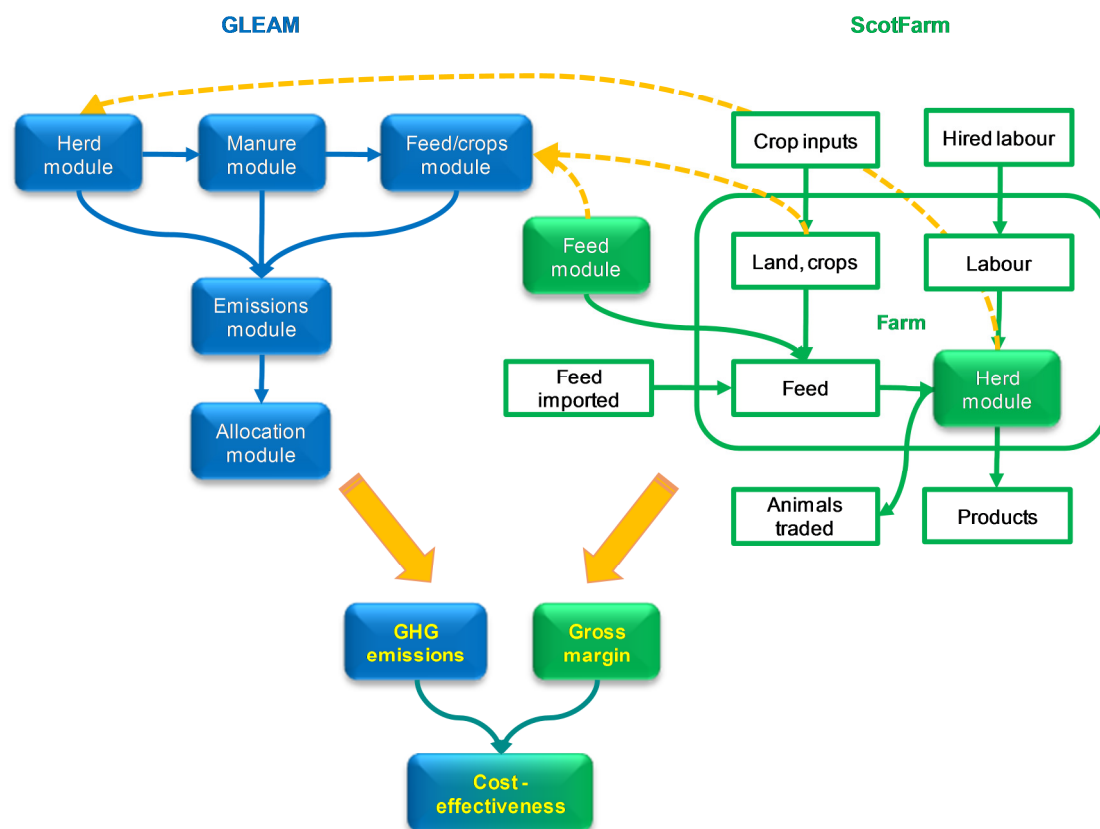


Figure 18. Conceptual framework of the linkage between GLEAM and ScotFarm

The main conceptual differences between the models are summarised in Table 18. To simulate both the baseline farms and the MMs in parallel in an optimisation and a static model, constraints are built in ScotFarm so that the farm structure of the baseline farm and the farm with the MM (apart from the specific changes due to the measure) is similar (i.e. the differences in grassland and arable land areas, herd size and feed composition between the farms modelled in GLEAM and in ScotFarm are not more than 5%). First the baseline farms are simulated in ScotFarm, and the resulting optimised baseline farm characteristics (land areas, number of cows, composition of the feed rations) are fed into GLEAM along with harmonised values for input parameters common to both models (e.g. milk and crop yields). The total production (of meat and milk) and GHG emissions are calculated in GLEAM and the farm gross margin is calculated in ScotFarm (see Figure 18). The procedure is then repeated for the scenario with the MM. The changes in emissions and in the EI of products due to the MM are then calculated by comparing the results of the baseline scenario and the scenario with the measure.

Table 18. Modelling differences between GLEAM and ScotFarm

	GLEAM	ScotFarm
Type of model	Static, deterministic calculation over 1 year	Linear programming pseudo dynamic optimisation model with yearly time-steps
System boundaries	Partial LCA: GHG emissions from cradle-to-delivery at retail point	Farm gate
Data input	Primary data such as animal numbers, herd/flock parameters, mineral fertiliser application rates, temperature, etc. derived sources such as literature, databases and surveys (see (MacLeod <i>et al.</i> 2013), Appendix B).	Farm level data such as land area, land use, animal numbers and labour use; and financial data such as gross margins, variable costs and overhead costs are taken from Farm Account Survey (Scottish Executive 2011). Farm coefficients such as livestock units and labour requirements are taken from The Farm Management Handbook (SAC 2012).
Output	Total annual commodity production (meat, milk and eggs); total GHG emissions; EI of each commodity.	Farm margins, feed rationing, herd size, land use
Dairy herd structure	Six animal categories based on reproductive use and sex, herd structure is calculated using herd parameters	Four animal categories based on age and sex; herd structure is optimised based on herd parameters and prices
Ration	Imported from ScotFarm	Endogenous – the financially optimal combination of feed materials that can meet nutritional constraints is determined. The nutritional constraints are the metabolisable energy and protein requirements based on age and production level of individual animals (Alderman and Cottrill 1993). Each of the farm groups however has to use concentrate diet at least 50% of level available in farm level data.

### 6.3.4 Defining farm types

Farm level data was drawn from the 2010/11 Farm Account Survey of Scotland (FAS) (Scottish Executive 2011). The FAS consisted of farm level data from 484 farms which included physical as well as financial information of each of the sampled farms. A cluster analysis was carried out in SPSS<sup>3</sup> to group farms together with similar characteristics. Seven farm variables (production system, farm gross margins, land, animal number, labour, feed and milk yield) were used to group the farms. These variables were assumed to be the main differences between farms. The Squared Euclidean Distance Method was used in finding similarities between the farms. This method is commonly used in cluster analysis when there are multi-dimensional variables such as farm variables used in this study (Solano *et al.* 2001).

<sup>3</sup> SPSS is a statistical software <http://www-01.ibm.com/software/analytics/spss/>

The cluster analysis resulted in fifteen farm types, with their main characteristics presented in Table 19. These characteristics formed the basis of more detailed farm descriptions, which were generated to describe the baseline farms in terms of their cropping and livestock activities, fertiliser and feed use, crops and livestock product yields.

*Table 19. Typology of Scottish farms generated, based on FAS*

Farm types	Grassland (ha)	Arable land (ha)	Livestock units <sup>1</sup> (lu)	Variable costs (€ lu <sup>-1</sup> )	Labour (man unit)
Dairy large	227.9	0.0	284	229.4	2.3
Dairy medium	99.5	11.7	137	227.7	2.1
Beef large	234.3	15.7	222	138.1	1.7
Beef medium	139.3	8.3	166	153.4	2.0
Beef small	77.0	4.5	84	143.0	1.3
Beef/Sheep large	263.5	27.9	242	151.2	2.9
Beef/Sheep medium	93.1	4.7	106	150.5	1.6
Sheep large	126.3	0.0	171	141.4	2.1
Sheep medium	65.3	0.0	81	126.0	1.5
Crop large	178.3	229.1	7	1428.6	7.5
Crop medium	86.3	218.0	8	1151.4	2.7
Crop small	46.6	89.0	3	1177.0	1.5
Mixed large	145.1	92.1	162	116.5	2.1
Mixed small	70.0	44.0	2045	112.5	1.6
Upland Beef/Sheep large	263.5	27.9	242	151.2	2.9
Upland Beef/Sheep medium	93.1	4.7	106	150.5	1.6
Low land Beef /Sheep	172.0	9.0	162	124.3	1.8

<sup>1</sup> Livestock unit: (defined in terms of feed requirement) one unit equals to the maintenance of a mature 625 kg Friesian cow and the production of a 40-45 kg calf and 4,500 l of milk per year

### 6.3.5 Case study: using sexed semen to reduce unwanted male calf numbers on Scottish dairy farms

In Scottish dairy herds, a proportion of the cows are mated, usually by artificial insemination, using dairy breed semen to produce replacement stock, and the remainder is inseminated with beef semen to provide dairy x beef calves that are reared for beef production. The use of unsexed semen leads to significant number of pure dairy male calves, most of which are not required for replacement, and may be uneconomic to rear as beef animals (Roberts *et al.* 2008). This raises issues of economic and resource inefficiency and animal welfare. The use of sexed semen enables the number of cows mated with dairy semen to be reduced and the number of dairy x beef calves to be increased (see Table 20). The effect of using sexed semen on the emissions arising from dairy production and on the farm finances were investigated.

*Table 20. Differences between baseline and MM assumptions*

Variable	Unsexed semen	Sexed semen
Proportion of female dairy replacement calves	0.35	0.35
Proportion of male dairy calves	0.35	0.05
Proportion of crossbred calves	0.30	0.60
Increase in the variable cost (€ lu <sup>-1</sup> )	-	11.7

Representing common practice in Scotland, the baseline farms were assumed to use artificial insemination, using dairy semen on 70% of their cows and heifers to produce enough female dairy calves for replacement (and as a ‘by-product’ dairy male calves, which are culled as newborns), and using beef semen on the remaining females to produce crossbred calves to be sold for rearing. With using sexed dairy semen the proportion of females inseminated with dairy semen is reduced to 40%, increasing the high-value crossbred calves proportion to 60%. The MM changes the income from the calves sold and the cost of the insemination in the financial model, and has effects on the GHG emissions from the reared beef cattle and on the meat produced.

The sexed semen mitigation method is only applicable on farms with dairy cattle: i.e. dairy and mixed farms, but it less relevant to mixed farms due to the much lower number of dairy cattle there, therefore the middle and large dairy farms were modelled. The main farm characteristics are presented in Table 21.

*Table 21. Main characteristics of the modelled baseline dairy farms*

Variable	Medium farm	Large farm
System: Year round calving, pasture based summer grazing for eight months, winter housing with grass silage feed, feed supplemented with concentrates and minerals year round.		
Number of cows (head)	149	300
Arable land area (ha)	11	0
Permanent grassland area (ha)	100	228
Milk sold (kg head <sup>-1</sup> year <sup>-1</sup> )	6000	7000
Milk price (€ l <sup>-1</sup> )	0.27	0.28
Crossbred calves price (€ head <sup>-1</sup> )	100	86
Cow weight (kg head <sup>-1</sup> )	540	
Fertility rate of cows	0.87	
Fertility rate of heifers	0.95	
Calving period	all year	
Calving interval (month)	12	
Age at first calving (month)	28	
Replacement rate	0.25	
Milk wastage ratio ( (milk secreted – milk sold) / milk secreted)	0.09	
Suckler beef EI (kg CO <sub>2</sub> e (carcass weight) <sup>-1</sup> )	30	



Two important parameters in the financial and EI reduction performance of the MM are the additional cost of using the sexed dairy semen and the assumption on the EI of the suckler beef the additional crossbred calves are replacing. Sensitivity analysis was undertaken to explore the influence of these assumptions on CE.

## 6.4 Results

Production, GHG emission and gross margin data of the baseline farms and the effect of using sexed semen are shown in Table 22. Producing more crossbred calves by using sexed semen increased the meat production of the systems by 47% for both medium and large dairy farms, while having no effect on milk production. This leads to an increase in the EI of the total protein produced, as a greater proportion of the protein is meat, which has a higher EI than milk. However, simply comparing the farms with and without sexed semen in term of the EI per unit of protein is misleading, as they are producing milk and meat in different proportions. In order to compare like with like, systems expansion can be used to isolate the emissions attributable to milk only. This is done by calculating the emissions that are avoided by producing beef, and subtracting these from the total emissions, to leave the emissions attributable to milk. In this example it is assumed that if the beef was not produced by the surplus dairy calves, it would have to be produced by specialised (i.e. cow-calf) beef production. This type of beef production typically has significantly higher EI than that of dairy beef (see (Opio *et al.* 2013) 2013, figure 12). It was assumed that the avoided specialised beef had an EI of 30 kg CO<sub>2</sub>e (kg carcass weight)<sup>-1</sup>. Under these assumptions, the EI of the milk is reduced by the MM by 9% and 12% on medium and large dairy farms, respectively. The financial modelling shows that the additional income from the increased number of marketable calves is more than 2.5 times more than the cost of sexed semen administration on both of the dairy farms. Therefore the CE of the measure on medium and large dairy farms is €-15 and €-7 (t CO<sub>2</sub>e)<sup>-1</sup>, respectively.

Table 22. Production, GHG emission and gross margin data (baseline and MM)

		Medium dairy farm		Large dairy farm	
		Baseline	With SS	Baseline	With SS
Production (kg protein year <sup>-1</sup> )	Meat	3,315	4,878	6,675	9,822
	Milk	29,591	29,591	68,815	68,815
GHG emissions for milk and meat (kg CO <sub>2</sub> e year <sup>-1</sup> )		2,144,750	2,366,120	4,559,644	5,005,356
EI of milk and meat protein (kg CO <sub>2</sub> e (kg protein) <sup>-1</sup> )		65.2	68.6	60.4	63.7
GHG emissions for milk only (kg CO <sub>2</sub> e year <sup>-1</sup> )		1,408,063	1,282,078	3,026,939	2,677,212
Milk EI (kg CO <sub>2</sub> e (kg milk) <sup>-1</sup> )		1.58	1.43	1.46	1.29
Gross margin (€ year <sup>-1</sup> )		165,284	167,128	261,569	264,120
Effect of MM		Medium dairy farm		Large dairy farm	
Change in milk GHG with sexed semen (kg CO <sub>2</sub> e year <sup>-1</sup> )		-125,984		-349,727	
Change in gross margin with sexed semen (€ year <sup>-1</sup> )		1,844		2,552	
CE of sexed semen (€ t CO <sub>2</sub> e <sup>-1</sup> )		-14.64		-7.30	

The sensitivity analysis shows that varying the EI of the suckler beef by +20% and -20% changes the abatement potential by +55% and -55%, respectively, while changing the variable cost increase (due to sexed semen administration) by +50% reduces the savings by 60% and increasing it by 100% or more makes a loss to the farmer. Overall, the CE of the measure varies between €-33 and €+27 t CO<sub>2</sub>e<sup>-1</sup> (Figure 19).

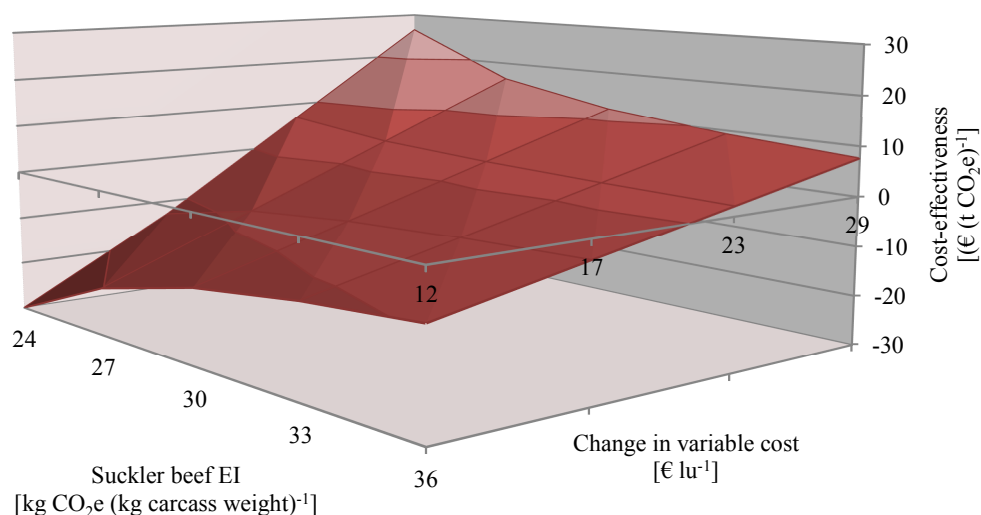


Figure 19. Sensitivity of the CE to the price of the sexed semen and the EI of the suckler beef

## 6.5 Discussion

Developing more efficient agro-environmental policies requires the CE of GHG MMs on different farm types to be quantified. The modelling framework proposed here provides this capacity, by using a financial optimisation model to simulate the gross margin changes and an LCA GHG model to estimate the emission changes arising from the MMs. Adopting an LCA-approach in these calculations helps to ensure that MMs do not simply displace emissions to other parts of the supply chain (although the danger of displacing production and emissions to other regions of the world still remains).

The current case study presents a Scottish dairy farm example; however, both GLEAM and ScotFarm have the flexibility to model a wide variety of farm types and locations, provided input data of the requisite type and quality is available. Further benefits of the framework are the consistency in assumptions across MMs and farm types and the inclusion of LCA and economic aspects to the whole farm modelling.

The modelling framework also has its limitations. The IPCC (2006) Tier 2 approach (IPCC 2006) to livestock and manure emissions used in GLEAM provides considerable scope for varying livestock parameters and, in so doing, the modelling of MMs. However, the Tier 1 approach to crop/soil emissions provides less scope (for example changes in the timing of fertiliser application or differences between soil types cannot be captured directly) and will be refined in the future versions of the model. The same applies to ScotFarm, where the cost breakdown distinguishes between labour, variable costs and overhead costs, therefore the MMs have to be described according to their effects on these variables rather than on more detailed farm activities. Nevertheless, these features also provide flexibility, as data collection at this level is quicker and often easier than acquiring farm type specific detailed activity and financial data. Therefore the results should be interpreted as for the ‘typical’ farm in the modelled region rather than specific to one individual farm. It is also important to mention that the current framework does not capture the co-effects of GHG mitigation on other pollutants. These effects – especially on other types of reactive N (e.g. ammonia and nitrate) – can be significant for some MMs, gaining even higher importance in regions with high nitrogen pollution. Nevertheless, these linked models provide a flexible and consistent way of calculating mitigation CE in a range of farm systems, helping to design better targeted regional and national policies for agriculture.

The results of the case study show that using sexed semen on dairy farms might be a cost-effective way (i.e. cheaper than the shadow price of carbon), in some circumstances even win-win opportunity (i.e. providing financial savings to the farmers) to reduce emissions from cattle production. An important aspect of this GHG mitigation is that the GHG savings do not occur directly on the farm. High-yielding, specialised dairy and beef systems are interlinked via the surplus calves in the dairy herds which can potentially be reared for meat and also via beef cross females from dairy herd becoming suckler cows. In the case of using sexed semen, the EI of the whole cattle system improves by decreasing the number of unwanted dairy male calves and increasing the amount of lower EI ‘dairy beef’ produced. The sensitivity analysis show that the measure stops generating financial savings on the farm after the additional cost of administering sexed semen exceeds approximately 21 € lu<sup>-1</sup>. Similarly, the GHG savings are highly sensitive to the assumption on the EI of the suckler beef production in the cattle system. The overall cost of sexed semen administration for the farmer depends not only on the cost of the semen but also on a number of factors related to fertility and herd management, like conception rate differences between cows and heifers, the availability of skilled personnel for the fertilisation, and the availability of sexed semen from high genetic merit sires. Providing more information and support in these areas to farmers would therefore increase the likelihood of the farmers achieving financial savings by using sexed semen in dairy herds. All in all, the feasibility of integrating sexed semen use into the Scottish Government’s GHG mitigation policy should be investigated.



## 7 Conclusions

Marginal abatement cost curves (MACCs) and cost-effectiveness assessments have proved to be popular instruments for informing environmental policy decisions. The usefulness of information provided by MACCs is maximised if users of this information are aware of the relevance and limitations of the analysis and, where possible, use alternative forms of MACCs, and complement their evaluation with other types of assessments, depending on the policy question in place.

This dissertation addressed five particular limitations of the MACC methodology. The frameworks developed were assessed in terms of whether they achieve the goal of providing more comprehensive information to policy makers than a conventional MACC.

Wider effects: the evidence presented here shows that

- It is possible to include the co-effects of greenhouse gas (GHG) mitigation measures in the cost-effectiveness calculations,
- Including the wider effects in the mitigation measures' cost-effectiveness can lead to changes in the economically optimal GHG abatement,
- There are important data gaps both in terms of the physical impacts and the monetary values associated with the pollutants, and
- In the case study presented, the highest damage cost values increased the economically efficient GHG abatement potential from 36.7% to 42.2% of the baseline emissions, while lower damage costs had no effect on it.

Transaction costs and the cost-effectiveness of policy instruments: the evidence presented here shows that

- The cost-effectiveness of policy instruments can be calculated with a modification of the MACC framework, where available transaction costs and uptake rates relevant to the policies in question are explicitly included,
- Transaction costs and uptake rates can prove to be difficult to estimate, and
- The case study reinforces that there is significantly reduced abatement potential when moving from technically feasible and cost-effective potential, to one that accommodates measures that are feasible to support via policies.

Uncertainty in the MACCs: the evidence presented here shows that

- There are numerous sources of uncertainty in the MACCs, and the qualitative

uncertainties can be assessed in a modified MACC framework,

- Reporting practices regarding uncertainty tend not to be rigorous and consistent, thus impeding the uncertainty assessment,
- In the case study reported the abatement rate and uptake are the most important input variables, and
- There is a high level of uncertainty regarding estimates of economically optimal abatement potential, but the ranking of mitigation measures and whether they are cost-effective or not is robust.

Boundaries of the analysis: the evidence presented here shows that

- The MACC modelling framework proposed here provides the capacity to adopt an life cycle assessment approach in the GHG calculations to distinguish between emission changes happening on the farm, and within the sector, and
- The results of the case study example show that using sexed semen on dairy farms might be a cost-effective way to reduce emissions from cattle production, though the GHG savings do not occur directly on the farm.

Heterogeneity: the evidence presented here shows that

- The modelling framework demonstrates a capacity to assess mitigation measures on different farm types, and
- The case study shows a higher improvement both in the gross margin and in the GHG emissions attributed to milk production on the large dairy farm than on the medium dairy farm. The improvement in the GHG emissions is relatively bigger, therefore the cost-effectiveness of using sexed semen is higher on the large dairy farm (in case of negative costs higher abatement results in higher cost-effectiveness).

A summary on how, in general, the limitations of the MACCs can affect the cost, cost-effectiveness and abatement estimates is offered in Table 23. This summary is intended for policy makers and other stakeholders and provides practical guidance on how to minimise these problems.

As presented in the examples in Table 2 and Table 23, and additionally via the case studies in this dissertation, agricultural MACCs and their information content can be improved so that to provide more tailored and comprehensive advice to policy makers. Some improvements are to be made in communication between scientists and stakeholders rather than in research methodology. This includes clearer definitions of the mitigation measures

and collaboration between all stakeholders to ensure that the scientific message translates into appropriate actions on farm. Choosing the cost-effectiveness threshold also requires a discussion between policy makers and researchers to ensure that the relevant marginal benefits are considered. Some potential limitations are already addressed in agricultural MACCs, widely, such as accounting for interactions, choice of discount rate, accounting for all main GHG effects. While methodological advances are not essential in these areas, future MACC analyses should consider these issues explicitly and provide details on how they addressed them. Methodologies and in some cases agricultural applications have been made available (including this dissertation) for other potential limitations, like boundaries of the analysis, heterogeneity, wider effects, transaction costs / policy cost-effectiveness and uncertainty. It is suggested that these methodological improvements should be applied more widely and relevant to the policy application.

Undeniably, the wider uptake of these approaches is hindered by multiple challenges. Lack of appropriate methodologies and data are amongst the most important challenges, for example methodological difficulties have so far prevented a robust estimation and inclusion of non-monetary barriers, while the lack of data results in estimating wider effects or transaction costs only partially. Importantly, the level of generalisation needed for a national bottom-up MACC requires data which are spatially and temporally averaged, therefore difficult to derive from experimental and modelling research. This generalisation problem is only enhanced when additional aspects are included in the analysis. MACC analysis is a highly applied assessment tool, very close to, and sometime directly embedded into the policy process. For this reason funding is often provided by governmental organisations for short and intensive projects, with an obvious consequence of simplified methodological approach. Additionally, reflecting the sometimes limited level of integration across policy objectives (e.g. climate change versus water quality, national targets versus global effects), the drive for expanded analysis might be weakened. Similarly, the desire of funders for certain type of analysis, e.g. uncertainty analysis, can be interfered by their preference of easily interpretable results which can give unambiguous answer to policy questions, even at the cost of partially losing the robustness of the results. Finally, traditional MACC analysis already requires collaboration between a wide range of scientific disciplines, which is often more challenging than non-interdisciplinary work. The expanded frameworks discussed above enhance this barrier by necessitating the close integration of further expertise, e.g. statisticians, ecologists or behavioural scientists.



Nevertheless, these barriers can be addressed in the medium to long term, benefiting not only MACC analyses, but other applied assessment methodologies. The methodological difficulties and data gaps are being reduced by emerging evidence, and the improvement can be accelerated by specifying and communicating particular research needs. Identifying research needs can also help in obtaining more robust evidence on generalised data, in this process targeted meta-analyses and synthesis research, along with appropriate use of expert elicitation have important roles. Directing some longer term research funds for MACC analysis and economic assessments would provide more opportunities for integrative research and for enhancing interdisciplinary capacities within and across research organisations and stakeholders. At the same time co-development of analytical frameworks between researchers and policy makers would enhance the understanding and use of more complex information in the policy process.

In the meantime quick further progress can be achieved with the more widespread application of the relatively less resource intensive methodological improvements, like creating a private and a social MACC using the relevant discount rate. Other improvements require the introduction of more complexity in the models (like addressing the problems around the boundaries of the analysis, wider effects, accounting for uncertainty), which, admittedly, could provide computational difficulties and impede the interpretation of the results. For short response time research a pragmatic approach is suggested therefore, whereby specific quantitative methodologies are used to answer the policy questions (see Table 23), and the results are complemented with qualitative analysis related to other limitations. Overall, MACCs are useful for informing the policy process, but should be used with full awareness of their limitations. Researchers providing MACC estimates are responsible for providing a clear indication of the important aspects of the MACCs. Without these pointers there is an increased likelihood of misinforming decision makers and designing inefficient policies. Here a guideline is suggested about the reporting of the MACC methodology. Following this guideline could facilitate future users' understanding of how the choice of methods affects the validity of the results. The following questions are proposed to be addressed in a summary section of MACC reports in the future.

- Boundaries
  - Are the results suitable more for sectoral/regional analysis or to look at the global GHG effects?
- Definitions of the mitigation measures
  - Provide technical details of the mitigation measures (What are the specific actions required from the farmers?)

- Discount rate
  - Private or social discount rate is used?
  - Which are the mitigation measures most affected by the choice of discount rate (i.e. measures with long lifetime)?
  - Are the assumptions on uptake aligned with the assumption on discount rate?
- GHG effects
  - What GHG emission sources are included?
- Heterogeneity
  - Is the limitation regarding heterogeneity addressed?
- Interactions
  - How are interactions regarding both emissions and costs included?
  - What is the basis for emission/mitigation calculation (IPCC, mass flow calculations, biophysical modelling), and how do these deal with interactions regarding e.g. nitrogen flow, livestock dietary options?
- Marginal benefits
  - To which year and region does the cost-effectiveness threshold used refer?
- Market effects
  - Are market effects considered and to what extent?
- Non-monetary barriers
  - Are any non-monetary barriers included, e.g. via the uptake rates?
- Transaction costs
  - Are any private or public transaction costs included?
- Wider effects
  - Are any wider effects included either quantitatively or qualitatively (e.g. through the pre-selection of mitigation measures)?
- Uncertainty
  - Has a sensitivity analysis been done? If yes, on which input data?
  - Has an uncertainty assessment been done?

Going forward it is important to keep in mind that cost-effectiveness and MACCs are able to explore and present important aspects of potential pollution reduction activities, but these aspects have to be complemented by other assessments. Furthermore, and most importantly, MACCs have to be embedded in a decision making process whereby all the important social, economic and environmental aspects are explored by the stakeholders.

Table 23. Summary of the main limitations of the MACC with suggested approach when providing information to policy decisions

Main limitations	Potential problems	Unitary cost-effectiveness <sup>a</sup>	Economically efficient mitigation	Suggested policy approach
<u>Boundaries</u> of the analysis are not fit for purpose or not clearly defined	If the boundaries are defined as the farm and/or domestic emissions, then there is a potential for emission leakage, i.e. some mitigation measures with seemingly low cost-effectiveness can be supported while they increase emissions outwith the farm gate or in other regions.	Over- or under-estimated	Over- or under-estimated	To assess which mitigation measures are the most cost-effective and provide the highest abatement at the global level use analysis looking at the whole supply chain and global effects. If not available, obtain quantitative assessment about the potential effects beyond the farm gate.
	If the boundaries are broader than the farm gate and/or domestic emissions and at the same time the analysis is cross-sectoral or covers multiple regions then part of the mitigation potential might be double counted.	Over- or under-estimated	Over- or under-estimated	To assess effort sharing between sectors or regions use MACCs which stay within the boundaries of the individual sectors/regions.
<u>Definitions of the mitigation measures</u> are not specific enough at the farm level	Communication between researchers, farmers, policy decision makers and other stakeholders is impeded. Actual changes in farming practices might differ from what had been suggested at the first place, likely reducing the mitigation effect.	No bias	No bias	Ensure that communication towards stakeholders is specific in articulating the suggested technical and management changes on farm.
<u>Discount rate</u> used is not fit for purpose	Private or public costs are under- or overestimated (especially for capital intensive mitigation measures), inter-generational equity is not addressed.	Over- or under-estimated	Over- or under-estimated	Use MACCs with contrasting (private and social) discount rates. If possible, use cost-effectiveness and uptake estimates where mitigation measures likely to be publicly/privately funded are assessed with a social/private discount rate, respectively.
<u>GHG effects</u> are not fully represented	Unintended emission or not-accounted mitigation might occur, thus the mitigation potential of some mitigation measures would be under- or overestimated.	Over- or under-estimated	Over- or under-estimated	Use MACCs considering changes in carbon stores (both soil and biomass) alongside nitrous oxide and methane, or, if not available, obtain quantitative assessment about which mitigation measures have potentially high effect on carbon stores.

<sup>a</sup> Note that the higher the cost-effectiveness the less favourable the mitigation measure is

Table 23. *cont.*

Main limitations	Potential problems	Unitary cost-effectiveness <sup>a</sup>	Economically efficient mitigation	Suggested policy approach
<u>Heterogeneity</u> is not represented	The differences in cost-effectiveness and mitigation potential between regions, farm types etc. are overlooked, therefore PIs might fail in some areas. Farms where the cost-effectiveness of a mitigation measure is higher than the average cost-effectiveness would show very low uptake.	No bias	Over-estimated	Use MACCs assessing heterogeneity in costs and abatement. If not available, flexible policy instruments can be designed to support mitigation measures with a wide range of private cost-effectiveness, e.g. linking financial support to expenses and opportunity costs occurred rather than providing a flat rate support.
<u>Interactions</u> between the mitigation measures and their effects on abatement and cost is not represented or not clearly defined	MACC is presented without accounting for interactions, therefore potentially double-counting part of the mitigation potential.	Under-estimated	Over-estimated	Only use MACCs which account for interactions when assessing total regional abatement. On the other hand, use cost-effectiveness and abatement of the individual mitigation measures (or a small package of mitigation measures) in regional policy design if realistically each individual farmer will not implement more than a few mitigation measures.
	The cost-effectiveness estimates calculated with considering interactions are used to assess cost-effectiveness in situations where the likely uptake would be limited to a few mitigation measures (therefore very limited interactions would occur).	Over-estimated	Under-estimated	
<u>Marginal benefits</u> are misrepresented	Incorrect cost-effectiveness threshold is used for defining the economically efficient mitigation	No bias	Over- or under-estimated	Ensure that the cost-effectiveness threshold used is consistent with the spatial and temporal relevance of the MACC; if possible, obtain sensitivity analysis results for the economically efficient mitigation at different thresholds.
<u>Market effects</u> are not represented in engineering and micro-economic MACCs	Potential effects on commodity markets are not captured and therefore some effects on food security, farm profitability and also on the cost-effectiveness of the mitigation measures might be overlooked.	Under-estimated	Over-estimated	Use general equilibrium ('top-down') MACCs to complement engineering ('bottom-up') MACCs, especially if large-scale changes in the amount of products or in farm finances is likely to happen.

<sup>a</sup> Note that the higher the cost-effectiveness the less favourable the mitigation measure is

Table 23. *cont.*

Main limitations	Potential problems	Unitary CE <sup>a</sup>	Economically efficient mitigation	Suggested policy approach
<u>Non-monetary barriers</u> are not represented	Voluntary uptake might be lower than predicted (and compulsory uptake might be more burdensome).	No bias	Over-estimated	Complement MACC analysis with analysis of the barriers of the different mitigation measures. Ideally, uptake estimates should take into account the potential main barriers in relation to each mitigation measure and policy instrument. Higher stakeholder involvement can help to improve the estimates.
<u>Transaction costs</u> and the cost-effectiveness of policies are not represented	If the private transaction costs are not captured, then the voluntary uptake might be lower than predicted (and compulsory uptake might be costlier). If the public transaction costs are not captured, then the cost of the policy instruments are underestimated, which might result in overspending or underdevelopment of the policy instruments .	Under-estimated	Over-estimated	If no MACC is available where transactions costs are estimated and built in, then use qualitative assessment of the likely level private and public transaction costs in relation to mitigation measures and PIs.
<u>Wider effects</u> are not represented	Integrated policy development is impeded: mitigation measures with negative co-effects might be supported, mitigation measures with positive co-effects might be overlooked.	No bias	Over- or under-estimated	Ensure that all the regionally/globally important environmental and societal effects are assessed, either included in the costs/benefits, or in physical terms. If no quantitative results are available, use a qualitative overview of the potential synergies and trade-offs.
<u>Uncertainty</u> is not represented	Not robust enough policy instruments , possible future changes in economic or climate/weather conditions can drastically reduce the cost-effectiveness of the policy instruments . If costs or baseline uptake underestimated or mitigation overestimated: costly or low mitigation measures might be overfunded, and <i>vice versa</i> .	Over- or under-estimated	Over- or under-estimated	Uncertainty analysis can be carried out on the MACC (e.g. Monte Carlo analysis). Alternatively robust decision making techniques can be used in policy development. Finally, sensitivity analysis can still reveal how the cost-effectiveness or the abatement potential might change in case of over- or underestimated input variables.

<sup>a</sup> Note that the higher the cost-effectiveness the less favourable the mitigation measure is

## References

- Anon. (2014) The Fourth Carbon Budget – reducing emissions through the 2020s, Committee on Climate Change.
- Aarnink, A. J. A., Swierstra, D., van den Berg, A. J. and Speelman, L. (1997) Effect of type of slatted floor and degree of fouling of solid floor on ammonia emission rates from fattening piggeries. *Journal of Agricultural Engineering Research* 66, 93-102.
- Aarnink, A. J. A. and Verstegen, M. W. A. (2007) Nutrition, key factor to reduce environmental load from pig production. *Livestock Science* 109, 194-203.
- Aberystwyth University (2010) Ruminant nutrition regimes to reduce methane and nitrogen emissions, Report No Defra AC0209, Defra, Institute of Biological, Environmental and Rural Sciences, Aberystwyth University, London.
- ADAS, EAE, AHDB, AIC, BPC, CLA, FWAG, LEAF, NFU, NIABTAG, Elm, F. and Royal Agricultural Society of England (2012) Meeting the challenge: Greenhouse Gas Action Plan Of the Agriculture Industry in England Progress report and Phase II Delivery.
- Akiyama, H., Yan, X. and Yagi, K. (2010) Evaluation of effectiveness of enhanced-efficiency fertilizers as mitigation options for N<sub>2</sub>O and NO emissions from agricultural soils: Meta-analysis. *Glob. Change Biol.* 16, 1837-1846.
- Alderman, G. and Cottrill, B. (1993) Energy and protein requirements of ruminants: an advisory manual prepared by the AFRC Technical Committee on Responses to Nutrients. An advisory manual prepared by the AFRC Technical Committee on responses to nutrients. CAB International Wallingford, UK.
- Alexandratos, N. and Bruinsma, J. (2012) World agriculture towards 2030/2050: the 2012 revision, Report No ESA Working Paper No. 12-03, Agricultural Development Economics Division, Food and Agriculture Organization of the United Nations.
- Amann, M., Bertok, I., Borken-Kleefeld, J., Cofala, J., Heyes, C., Hoglund-Isaksson, L., Klimont, Z., Nguyen, B., Posch, M., Rafaj, P., Sandler, R., Schopp, W., Wagner, F. and Winiwarter, W. (2011) Cost-effective control of air quality and greenhouse gases in Europe: Modeling and policy applications. *Environmental Modelling & Software* 26, 1489-1501.
- Amon, B., Amon, T., Boxberger, J. and Alt, C. (2001) Emissions of NH<sub>3</sub>, N<sub>2</sub>O and CH<sub>4</sub> from dairy cows housed in a farmyard manure tying stall (housing, manure storage, manure spreading). *Nutrient Cycling in Agroecosystems* 60, 103-113.
- Amon, B., Boxberger, J., Amon, T., Zaussinger, A. and Pollinger, A. (1997) Emission data of NH<sub>3</sub>, CH<sub>4</sub> and N<sub>2</sub>O from fattening bulls, milking cows and during different ways of storing solid manure. Symposium on Ammonia and Odour Emissions from Animal Production Facilities, Vinkeloord, The Netherlands, 6-10 October 1997, Proceedings of the International Symposium on Ammonia and Odour Control from Animal Production Facilities, J. A. M. Voermans & G. J. Monteny, eds., Research Institute for Pig Husbandry (Proefstation voor de Varkenshouderij), Rosmalen.

- Amon, B., Kryvoruchko, V., Amon, T. and Zechmeister-Boltenstern, S. (2006) Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. *Agriculture, Ecosystems & Environment* 112, 153-162.
- Amon, B., Kryvoruchko, V., Frohlich, M., Amon, T., Pollinger, A., Mosenbacher, I. and Hausleitner, A. (2007) Ammonia and greenhouse gas emissions from a straw flow system for fattening pigs: Housing and manure storage. *Livestock Science* 112, 199-207.
- Andreas Schafer and Henry (2006) Experiments with a hybrid CGE-MARKAL model. *The Energy Journal Hybrid Modeling*, 171-178.
- Anthoff, D., Hepburn, C. and Tol, R. S. J. (2009) Equity weighting and the marginal damage costs of climate change. *Ecological Economics* 68, 836-849.
- Anthoff, D. and Tol, R. S. J. (2010) On international equity weights and national decision making on climate change. *Journal of Environmental Economics and Management* 60, 14-20.
- Anthony, S., Duethman, D., Gooday, R., Harris, D., Newell-Price, P., Chadwick, D. and Misselbrook, T. (2008) Quantitative assessment of scenarios for managing trade-off between economics, environment and media, Report No Defra WQ0106 (Module 6), Defra, ADAS, North Wyke Research.
- Arrow, K., Cropper, M., Gollier, C., Groom, B., Heal, G., Newell, R., Nordhaus, W., Pindyck, R., Pizer, W., Portney, P., Sterner, T., Tol, R. S. J. and Weitzman, M. (2013) Determining Benefits and Costs for Future Generations. *Science* 341, 349-350.
- Audit Scotland (2011) Reducing Scottish greenhouse gas emissions, Audit Scotland.
- Baker, B., Metcalfe, P., Butler, S., Gueron, Y., Sheldon, R. and East, J. (2007) The benefits of water framework directive programmes of measures in England and Wales, Report No Defra CRP 4b/c, Defra.
- Beauchemin, K., Kreuzer, M., O'Mara, F. and McAllister, T. (2008) Nutritional management for enteric methane abatement: A review. *Aust. J. Exp. Agric.* 48, 21-27.
- Beauchemin, K. A., Henry Janzen, H., Little, S. M., McAllister, T. A. and McGinn, S. M. (2010) Life cycle assessment of greenhouse gas emissions from beef production in western Canada: A case study. *Agricultural Systems* 103, 371-379.
- Beddington, J. R., Asaduzzaman, M., Clark, M. E., Fern+índez Bremauntz, A., Guillou, M. D., Howlett, D. J. B., Jahn, M. M., Lin, E., Mamo, T., Negra, C., Nobre, C. A., Scholes, R. J., Van Bo, N. and Wakhungu, J. (2012) What next for agriculture after Durban? *Science* 335, 289-290.
- Bellarby, J., Tirado, R., Leip, A., Weiss, F., Lesschen, J. P. and Smith, P. (2013) Livestock greenhouse gas emissions and mitigation potential in Europe. *Glob. Change Biol.* 19, 3-18.
- Berg, W. and Hornig, G. (1997) Emission reduction by acidification of slurry investigations and assessment. Symposium on Ammonia and Odour Emissions from Animal Production Facilities, Vinkeloord, The Netherlands, 6-10 October 1997, Proceedings of the International Symposium on Ammonia and Odour Control from Animal Production Facilities, J. A. M.

Voermans & G. J. Monteny, eds., Research Institute for Pig Husbandry (Proefstation voor de Varkenshouderij), Rosmalen.

Berg, W., Brunsch, R. and Pazsiczki, I. (2006) Greenhouse gas emissions from covered slurry compared with uncovered during storage. *Agriculture, Ecosystems & Environment* 112, 129-134.

Biggar, S., Man, D., Moffroid, K., Pape, D., Riley-Gilbert, M., Steele, R. and Thompson, V. (2013) Greenhouse gas mitigation options and costs for agricultural land and animal production within the United States, Report No Contract No. AG-3142-P-10-0214, ICF International, U.S. Department of Agriculture Climate Change Program Office Washington, DC.

Blok, K., de Jager, D., Hendriks, C., Kouvaritakis, N. and Mantzos, L. (2001) Economic Evaluation of Sectoral Emission Reduction Objectives for Climate Change, European Commission.

Bouwman, A. F. (1998) Nitrogen oxides and tropical agriculture. *Nature* 392, 866-867.

Bouwmeester, R. J. B. and Vlek, P. L. G. (1981) Rate control of ammonia volatilization from rice paddies. *Atmospheric Environment* (1967) 15, 131-140.

Bremner, J. M. and Blackmer, A. M. (1978) Nitrous oxide - Emission from soils during nitrification of fertilizer nitrogen. *Science* 199, 295-296.

Brink, C., van Grinsven, H., Jacobsen, B. H., Rabl, A., Gren, I. M., Holland, M., Klimont, Z., Hicks, W. K., Brouwer, R., Dickens, R., Willems, J., Termansen, M., Velthof, G. L., Alkemade, R., van Oorschot, M. and Webb, J. (2011) Costs and benefits of nitrogen in the environment, In: edited by M. A. Sutton, C. M. Howard, J. W. Erisman, G. Billen, A. Bleeker, P. Grennfelt, H. Grinsven, & B. Grizzetti (ed) *The European Nitrogen Assessment*, First Cambridge University Press, Cambridge, pp. 513-540.

Brink, C., Kroeze, C. and Klimont, Z. (2001) Ammonia abatement and its impact on emissions of nitrous oxide and methane--Part 2: application for Europe. *Atmospheric Environment* 35, 6313-6325.

Brink, C., van Ierland, E., Hordijk, L. and Kroeze, C. (2005) Cost-effective emission abatement in agriculture in the presence of interrelations: cases for the Netherlands and Europe. *Ecological Economics* 53, 59-74.

Broderick, G. A., Stevenson, M. J., Patton, R. A., Lobos, N. E. and Olmos Colmenero, J. J. (2008) Effect of supplementing rumen-protected methionine on production and nitrogen excretion in lactating dairy cows. *J Dairy Sci* 91, 1092-1102.

Brouwer, R. (2000) Environmental value transfer: state of the art and future prospects. *Ecological Economics* 32, 137-152.

Bruce, J. M. (1990) Straw-flow - A high welfare system for pigs. *Farm Building Progress* 9-13.

Buckingham, S., Anthony, S., Bellamy, P. H., Cardenas, L. M., Higgins, S., McGeough, K. and Topp, C. F. E. (2014) Review and analysis of global agricultural N<sub>2</sub>O emissions relevant to the UK. *Science of the Total Environment* 487, 164-172.



Bussink, D. W. and Oenema, O. (1998) Ammonia volatilization from dairy farming systems in temperate areas: a review. *Nutrient Cycling in Agroecosystems* 51, 19-33.

Canh, T. T., Aarnink, A. J. A., Schutte, J. B., Sutton, A., Langhout, D. J. and Verstegen, M. W. A. (1998a) Dietary protein affects nitrogen excretion and ammonia emission from slurry of growing-finishing pigs. *Livestock Production Science* 56, 181-191.

Canh, T. T., Aarnink, A. J. A., Verstegen, M. W. A. and Schrama, J. W. (1998b) Influence of dietary factors on the pH and ammonia emission of slurry from growing-finishing pigs. *Journal of Animal Science* 76, 1123-1130.

Canh, T. T., Verstegen, M. W. A., Aarnink, A. J. A. and Schrama, J. W. (1997) Influence of dietary factors on nitrogen partitioning and composition of urine and feces of fattening pigs. *Journal of Animal Science* 75, 700-706.

Carmona, G., Christianson, C. B. and Byrnes, B. H. (1990) Temperature and low concentration effects of the urease inhibitor N-(n-butyl) thiophosphoric triamide (nBTPT) on ammonia volatilization from urea. *Soil Biology and Biochemistry* 22, 933-937.

Chadwick, D., Misselbrook, T., Gilhespy, S., Williams, J., Bhogal, A., Sagoo, L., Nicholson, F., Webb, J., Anthony, S. and Chambers, B. (2005) Ammonia emissions and crop N use efficiency to grassland and tillage land; Factors affecting ammonia emissions from urea based fertilisers; Ammonia emissions model, Report No Defra NT2605 WP1b, Defra, IGER, ADAS.

Chadwick, D., Sommer, S., Thorman, R., Fangueiro, D., Cardenas, L., Amon, B. and Misselbrook, T. (2011) Manure management: Implications for greenhouse gas emissions. *Animal Feed Science and Technology* 166-67, 514-531.

Chadwick, D. R. (2005) Emissions of ammonia, nitrous oxide and methane from cattle manure heaps: effect of compaction and covering. *Atmospheric Environment* 39, 787-799.

Chambers, B. J., Lord, E. I., Nicholson, F. A. and Smith, K. A. (1999) Predicting nitrogen availability and losses following application of organic manures to arable land: MANNER. *Soil Use Manage* 15, 137-143.

Clemens, J., Trimborn, M., Weiland, P. and Amon, B. (2006) Mitigation of greenhouse gas emissions by anaerobic digestion of cattle slurry. *Agriculture, Ecosystems & Environment* 112, 171-177.

Coggan, A., Whitten, S. M. and Bennett, J. (2010) Influences of transaction costs in environmental policy. *Ecological Economics* 69, 1777-1784.

Collier, U., Kennedy, D., Bellamy, O., Gault, A., Hafraoui, H., Meddings, N. and Srinivasan, K. (2013a) Reducing the UK's carbon footprint, Committee on Climate Change.

Collier, U., Kennedy, D., Kmietowicz, E., Bellamy, O., Gault, A., Hafraoui, H., Hill, J., Kazaglis, A., Meddings, N., Srinivasan, K. and Thillainathan, I. (2013b) Reducing the UK's carbon footprint and managing competitiveness risks, Committee on Climate Change.

Correll, D. L. (1998) The Role of Phosphorus in the Eutrophication of Receiving Waters: A Review. *J. Environ. Qual.* 27, 261-266.

- Cottle, D. J., Nolan, J. V. and Wiedemann, S. G. (2011) Ruminant enteric methane mitigation: a review. *Anim. Prod. Sci.* 51, 491-514.
- Crosson, P., Shalloo, L., O'Brien, D., Lanigan, G. J., Foley, P. A., Boland, T. M. and Kenny, D. A. (2011) A review of whole farm systems models of greenhouse gas emissions from beef and dairy cattle production systems. *Animal Feed Science and Technology* 166-167, 29-45.
- Cuttle, S., Shepherd, M. A., Lord, E. I. and Hillman, J. (2004) Literature review of the effectiveness of measures to reduce nitrate leaching from agricultural land, Report No Defra NT2511, Defra, IGER, ADAS.
- Dalgaard, T., Bienkowski, J. F., Bleeker, A., Dragosits, U., Drouet, J. L., Durand, P., Frumau, A., Hutchings, N. J., Kedziora, A., Magliulo, V., Olesen, J. E., Theobald, M. R., Maury, O., Akkal, N. and Cellier, P. (2012) Farm nitrogen balances in six European landscapes as an indicator for nitrogen losses and basis for improved management. *Biogeosciences* 9, 5303-5321.
- Dalgaard, T., Hutchings, N., Dragosits, U., Olesen, J. E., Kjeldsen, C., Drouet, J. L. and Cellier, P. (2011) Effects of farm heterogeneity and methods for upscaling on modelled nitrogen losses in agricultural landscapes. *Environmental Pollution* 159, 3183-3192.
- Davison, P. S., Withers, P. J., Lord, E. I., Betson, M. J. and Stroemqvist, J. (2008) PSYCHIC - A process-based model of phosphorus and sediment mobilisation and delivery within agricultural catchments. Part 1: Model description and parameterisation. *Journal of Hydrology* 350, 290-302.
- Dawar, K., Zaman, M., Rowarth, J., Blennerhassett, J. and Turnbull, M. (2011) Urease inhibitor reduces N losses and improves plant-bioavailability of urea applied in fine particle and granular forms under field conditions. *Agriculture Ecosystems & Environment* 144, 41-50.
- de Bruin, K., Dellink, R. B., Ruijs, A., Bolwidt, L., van Buuren, A., Graveland, J., de Groot, R. S., Kuikman, P. J., Reinhard, S., Roetter, R. P., Tassone, V. C., Verhagen, A. and van Ierland, E. C. (2009) Adapting to climate change in the Netherlands: an inventory of climate adaptation options and ranking of alternatives. *Climatic Change* 95, 23-45.
- De Cara, S., Houze, M. and Jayet, P. A. (2005) Methane and nitrous oxide emissions from agriculture in the EU: A spatial assessment of sources and abatement costs. *Environmental & Resource Economics* 32, 551-583.
- De Cara, S. and Jayet, P. A. (2000) Emissions of greenhouse gases from agriculture: The heterogeneity of abatement costs in France. *European Review of Agricultural Economics* 27, 281-303.
- DeAngelo, B. J., de la Chesnaye, F. C., Beach, R. H., Sommer, A. and Murray, B. C. (2006) Methane and nitrous oxide mitigation in agriculture. *Energy Journal* 27, 89-108.
- DECC (2009) Carbon Valuation in UK Policy Appraisal: A Revised Approach, Climate Change Economics, Department of Energy and Climate Change (DECC), London.
- Defra: Damage cost estimates  
<http://archive.defra.gov.uk/environment/quality/air/airquality/panels/igcb/guidance/damagecosts.htm> Accessed: 22-6-2011.

Defra (2011) Greenhouse gas emission projections for UK agriculture to 2030, Report No PB 13622, Defra, London.

Dewes, T. (1996) Effect of pH, temperature, amount of litter and storage density on ammonia emissions from stable manure. *Journal of Agricultural Science* 127, 501-509.

Dijkstra, J., Oenema, O. and Bannink, A. (2011) Dietary strategies to reducing N excretion from cattle: implications for methane emissions. *Current Opinion in Environmental Sustainability* 3, 414-422.

Dinuccio, E., Berg, W. and Balsari, P. (2008) Gaseous emissions from the storage of untreated slurries and the fractions obtained after mechanical separation. *Atmospheric Environment* 42, 2448-2459.

Dinuccio, E., Berg, W. and Balsari, P. (2011) Effects of mechanical separation on GHG and ammonia emissions from cattle slurry under winter conditions. *Animal Feed Science and Technology* 166-167, 532-538.

Dominguez, I. P., Britz, W. and Holm-Muller, K. (2009) Trading schemes for greenhouse gas emissions from European agriculture: A comparative analysis based on different implementation options. *Review of Agricultural and Environmental Studies - Revue d'Etudes en Agriculture et Environnement* 90, 287-308.

Ducos, G., Dupraz, P. and Bonnieux, F. (2009) Agri-environment contract adoption under fixed and variable compliance costs. *Journal of Environmental Planning and Management* 52, 669-687.

Edwards, S., Edge, H., Cain, P., Guy, J., Hazzledine, M. and Gill, P. (2002) A review and workshop to agree advice on reduced nutrient pig diets - Final Report, Report No Defra WA0321, Defra, University of Newcastle upon Tyne, London.

El Kader, N. A., Robin, P., Paillat, J. M. and Leterme, P. (2007) Turning, compacting and the addition of water as factors affecting gaseous emissions in farm manure composting. *Bioresource Technology* 98, 2619-2628.

Ellis, J., Dijkstra, J., Kebreab, E., Bannink, A., Odongo, N., McBride, B. and France, J. (2008) Aspects of rumen microbiology central to mechanistic modelling of methane production in cattle. *Journal of Agricultural Science* 146, 213-233.

Ellis, J. L., Dijkstra, J., Bannink, A., Parsons, A. J., Rasmussen, S., Edwards, G. R., Kebreab, E. and France, J. (2011) The effect of high-sugar grass on predicted nitrogen excretion and milk yield simulated using a dynamic model. *J Dairy Sci* 94, 3105-3118.

Enkvist, P. A., Naucler, T. and Rosander, J. (2007) A cost curve for greenhouse gas reduction McKinsey.

EPA (2012) Global anthropogenic non-CO<sub>2</sub> greenhouse gas emissions: 1990-2030, Report No EPA 430-R-12-006, United States Environmental Protection Agency, Office of Atmospheric Programs, Washington DC.

Eugene, M., Masse, D., Chiquette, J. and Benchaar, C. (2008) Meta-analysis on the effects of lipid supplementation on methane production in lactating dairy cows. *Can. J. Anim. Sci.* 88, 331-337.

European Environment Agency (2013) European Union emission inventory report 1990–2011 under the UNECE Convention on Long-range Transboundary Air Pollution (LRTAP), Report No EEA Technical Report No 10/2013, European Environment Agency.

European Environment Agency (2014) Annual European Union greenhouse gas inventory 1990-2012 and inventory report 2014, Report No EEA Technical report No 9/2014, European Environment Agency, Copenhagen.

Fabbri, C., Valli, L., Guarino, M., Costa, A. and Mazzotta, V. (2007) Ammonia, methane, nitrous oxide and particulate matter emissions from two different buildings for laying hens. *Biosystems Engineering* 97, 441-455.

Fangueiro, D., Coutinho, J., Chadwick, D., Moreira, N. and Trindade, H. (2008) Effect of cattle slurry separation on greenhouse gas and ammonia emissions during storage. *J. Environ. Qual.* 37, 2322-2331.

Fangueiro, D., Ribeiro, H., Coutinho, J., Cardenas, L., Trindade, H., Cunha-Queda, C., Vasconcelos, E. and Cabral, F. (2010) Nitrogen mineralization and CO<sub>2</sub> and N<sub>2</sub>O emissions in a sandy soil amended with original or acidified pig slurries or with the relative fractions. *Biology and Fertility of Soils* 46, 383-391.

Feliciano, D., Hunter, C., Slee, B. and Smith, P. (2014) Climate change mitigation options in the rural land use sector: Stakeholders' perspectives on barriers, enablers and the role of policy in North East Scotland. *Environmental Science & Policy* 44, 26-38.

Firestone, M. K., Firestone, R. B. and Tiedje, J. M. (1980) Nitrous oxide from soil denitrification: Factors controlling its biological production. *Science* 208, 749-751.

Foresight (2011) The Future of Food and Farming: Challenges and choices for global sustainability. Final Project Report, The Government Office for Science, London.

Frankhauser, S. and Tol, R. S. (1996) Climate change costs: Recent advancements in the economic assessment. *Energy Policy* 24, 665-673.

Fukumoto, Y., Osada, T., Hanajima, D. and Haga, K. (2003) Patterns and quantities of NH<sub>3</sub>, N<sub>2</sub>O and CH<sub>4</sub> emissions during swine manure composting without forced aeration - effect of compost pile scale. *Bioresource Technology* 89, 109-114.

Galloway, J. N., Aber, J. D., Erisman, J. W., Seitzinger, S. P., Howarth, R. W., Cowling, E. B. and Cosby, B. J. (2003) The nitrogen cascade. *BioScience* 53, 341-356.

Georgiou, S., Bateman, I., Cole, M. and Hadley, D. (2000) Contingent ranking and valuation of river water quality improvements: testing for scope sensitivity, ordering and distance decay effects, Report No CSERGE Working Paper GEC 2000-18.

Gerber, P., Vellinga, T., Dietze, K., Falcucci, A., Gianni, G., Mounsey, J., Maiorano, L., Opio, C., Sironi, D., Thieme, O., . and Weiler, V. (2010) Greenhouse Gas Emissions from the Dairy Sector - A Life Cycle Assessment, FAO, Rome.

Gibbons, J. M., Ramsden, S. J. and Blake, A. (2006) Modelling uncertainty in greenhouse gas emissions from UK agriculture at the farm level. *Agriculture Ecosystems & Environment* 112, 347-355.

- Gielen, D. and Changhong, C. (2001) The CO<sub>2</sub> emission reduction benefits of Chinese energy policies and environmental policies: A case study for Shanghai, period 1995–2020. *Ecological Economics* 39, 257-270.
- Gilhespy, S. L., Webb, J., Chadwick, D. R., Misselbrook, T. H., Kay, R., Camp, V., Retter, A. L. and Bason, A. (2009) Will additional straw bedding in buildings housing cattle and pigs reduce ammonia emissions? *Biosystems Engineering* 102, 180-189.
- Glenk, K. and Colombo, S. (2011) Designing policies to mitigate the agricultural contribution to climate change: an assessment of soil based carbon sequestration and its ancillary effects. *Climatic Change* 105, 43-66.
- Golub, A., Hertel, T., Lee, H. L., Rose, S. and Sohngen, B. (2009) The opportunity cost of land use and the global potential for greenhouse gas mitigation in agriculture and forestry. *Resource and Energy Economics* 31, 299-319.
- Grainger, C. and Beauchemin, K. A. (2011) Can enteric methane emissions from ruminants be lowered without lowering their production? *Animal Feed Science and Technology* 166-167, 308-320.
- Gren, I. M., Jonzon, Y. and Lindqvist, M. (2008) Costs of nutrient reductions to the Baltic Sea. Technical report, Report No Working Paper Series 2008:1, Swedish University of Agricultural Sciences (SLU), Department of Economics, Uppsala.
- Grijalva, T., Lusk, J. and Shaw, W. D. (2014) Discounting the distant future: An experimental investigation. *Environ Resource Econ* 59, 39-63.
- Groot, J. C. J., Rossing, W. A. H. and Lantinga, E. A. (2006) Evolution of farm management, nitrogen efficiency and economic performance on Dutch dairy farms reducing external inputs. *Livestock Science* 100, 99-110.
- Hallegatte, S., Shah, A., Lempert, R., Brown, C. and Gill, S. (2012) Investment Decision Making under Deep Uncertainty — Application to Climate Change, Report No WPS6193, The World Bank Sustainable Development Network.
- Halvorson, A. D., Del Grosso, S. J. and Alluvione, F. (2010) Nitrogen source effects on nitrous oxide emissions from irrigated no-till corn. *J. Environ. Qual.* 39, 1554-1562.
- Halvorson, A. D., Del Grosso, S. J. and Jantalia, C. P. (2011) Nitrogen source effects on nitrous oxide emissions from strip-till corn. *J. Environ. Qual.* 40, 1775-1786.
- Hansen, M. N., Henriksen, K. and Sommer, S. G. (2006) Observations of production and emission of greenhouse gases and ammonia during storage of solids separated from pig slurry: Effects of covering. *Atmospheric Environment* 40, 4172-4181.
- Harris, D., Jones, G., Elliott, J., Williams, J., Chambers, B. J. and Dyer, R. (2009) Analysis of policy instruments for reducing greenhouse gas emissions from agriculture, forestry and land management, Report No Defra RMP/5142, Defra, ADAS, London.
- Harrison, R. and Webb, J. (2001) A review of the effect of N fertilizer type on gaseous emissions. *Advances in Agronomy* 73, 65-108.

Hartung, J. and Phillips, V. R. (1994) Control of gaseous emissions from livestock buildings and manure stores. *Journal of Agricultural Engineering Research* 57, 173-189.

Hassouna, M., Espagnol, S., Robin, P., Paillat, J. M., Levasseur, P. and Li, Y. (2008) Monitoring NH<sub>3</sub>, N<sub>2</sub>O, CO<sub>2</sub> and CH<sub>4</sub> emissions during pig solid manure storage - Effect of turning. *Compost Science & Utilization* 16, 267-274.

Haygarth, P., ApSimon, H., Betson, M., Harris, D., Hodgkinson, R. and Withers, P. (2009) Mitigating Diffuse Phosphorus Transfer from Agriculture According to Cost and Efficiency. *J. Environ. Qual.* 38, 2012-2022.

Heijungs, R. (1996) Identification of key issues for further investigation in improving the reliability of life-cycle assessments. *Journal of Cleaner Production* 4, 159-166.

Hellmann, B., Zelles, L., Palojarvi, A. and Bai, Q. Y. (1997) Emission of climate-relevant trace gases and succession of microbial communities during open-window composting. *Applied and Environmental Microbiology* 63, 1011-1018.

Hennessy, T., Shrestha, S. and Farrell, M. (2008) Quantifying the viability of farming in Ireland: can decoupling address the regional imbalances? *Irish Geography* 41, 29-47.

Holland, M., Pye, S., Watkiss, P., Droste-Franke, B. and Bickel, P. (2005) Damages per tonne emission of PM<sub>2.5</sub>, NH<sub>3</sub>, SO<sub>2</sub>, NO<sub>x</sub> and VOCs from each EU25 Member State (excluding Cyprus) and surrounding seas, Report No Clean Air for Europe (CAFE) Programme, AEA Technology Environment; European Commission DG Environment.

Hristov, A., Hanigan, M., Cole, A., Todd, R., McAllister, T., Ndegwa, P. and Rotz, A. (2011) Review: Ammonia emissions from dairy farms and beef feedlots. *Can. J. Anim. Sci.* 91, 1-35.

Hristov, A., Oh, J., Lee, C., Meinen, R., Montes, F., Ott, T., Firkins, J., Rotz, A., Dell, C., Adesogan, A., Yang, W. Z., Tricarico, J., Kebreab, E., Waghorn, G. C., Dijkstra, J. and Oosting, S. Gerber, P., Henderson, B., and Makkar, H. (ed) (2013) Mitigation of greenhouse gas emissions in livestock production - A review of technical options for non-CO<sub>2</sub> emissions, Report No FAO Animal Production and Health Paper 177, FAO, Rome, Italy.

Hudson, N., Duperouzel, D. and Melvin, S. (2006) Assessment of permeable covers for odour reduction in piggery effluent ponds. 1. Laboratory-scale trials. *Bioresource Technology* 97, 2002-2014.

Hutchings, N. J., Sommer, S. G., Andersen, J. M. and Asman, W. A. H. (2001) A detailed ammonia emission inventory for Denmark. *Atmospheric Environment* 35, 1959-1968.

IGER (2001) Cost curve assessment of mitigation options in greenhouse gas emissions from agriculture, Report No Defra CC0229, Defra, London.

IGER (2005) Evaluation of targeted or additional straw use as a means of reducing NH<sub>3</sub> emissions from buildings housing cattle and pigs, Report No Defra AM0103, Defra, London.

IPCC (1997) Revised 1996 IPCC guidelines for national greenhouse gas inventories, IPCC WGI Technical Support Unit, Hadley Centre, Meteorological Office, Bracknell, UK.

IPCC (2000) Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories, IPCC National Greenhouse Gas Inventories Programme, Technical Support Programme Technical Support Unit, Institute for Global Environmental Strategies, Hayama, Kanagawa, Japan.

IPCC Eggleston, H. S., Buendia, L., Miwa, K., Ngara, T., and Tanabe, K. (ed) (2006) 2006 IPCC guidelines for national greenhouse gas inventories, Prepared by the National Greenhouse Gas Inventories Programme, Volume 4: Agriculture, forestry and other land use, Institute for Global Environmental Strategies (IGES), Japan.

IPCC Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K. B., Tignor, M., and Miller, H. L. (ed) (2007) Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

IPCC Stocker, T. F., Qin, D., Plattner, G. K., Tignor, M., Allen, S. K., Boschung, J., Nauels, A., Xia, Y., Bex, V., and Midgley, P. M. (ed) (2013a) Climate change 2013: The physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

IPCC (2013b) Summary for policymakers, In: edited by T. F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex, & P. M. Midgley (ed) Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, WGI AR5 PM Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1-36.

Johnson, K. A. and Johnson, D. E. (1995) Methane emissions from cattle. *Journal of Animal Science* 73, 2483-2492.

Jonker, J. S., Kohn, R. A. and High, J. (2002) Dairy herd management practices that impact nitrogen utilization efficiency. *J Dairy Sci* 85, 1218-1226.

Jungbluth, T., Hartung, E. and Brose, G. (2001) Greenhouse gas emissions from animal houses and manure stores. *Nutrient Cycling in Agroecosystems* 60, 133-145.

Kai, P., Pedersen, P., Jensen, J. E., Hansen, M. N. and Sommer, S. G. (2008) A whole-farm assessment of the efficacy of slurry acidification in reducing ammonia emissions. *European Journal of Agronomy* 28, 148-154.

Kann, A. and Weyant, J. (2000) Approaches for performing uncertainty analysis in large-scale energy/economic policy models. *Environmental Modeling & Assessment* 5, 29-46.

Kasterine, A. and Vanzetti, D. (2010) The effectiveness, efficiency and equity of market based and voluntary measures to mitigate greenhouse gas emissions from the agri-food sector. *Trade and Environment Review 2010*, United Nations Conference on Trade and Development (UNCTAD), Geneva,

Kesicki, F. and Ekins, P. (2012) Marginal abatement cost curves: a call for caution. *Climate Policy* 12, 219-236.

- Kesicki, F. and Strachan, N. (2011) Marginal abatement cost (MAC) curves: confronting theory and practice. *Environmental Science & Policy* 14, 1195-1204.
- Khan, R. Z., Muller, C. and Sommer, S. G. (1997) Micrometeorological mass balance technique for measuring CH<sub>4</sub> emission from stored cattle slurry. *Biology and Fertility of Soils* 24, 442-444.
- Kim, I. B., Ferket, P. R., Powers, W. J., Stein, H. H. and van Kempen, T. A. T. G. (2004) Effects of different dietary acidifier sources of calcium and phosphorus on ammonia, methane and odorant emission from growing-finishing pigs. *Asian-Australasian Journal of Animal Sciences* 17, 1131-1138.
- Kirchmann, H. and Witter, E. (1989) Ammonia volatilization during aerobic and anaerobic manure decomposition. *Plant and Soil* 115, 35-41.
- Knaggard, A. (2013) What do policy-makers do with scientific uncertainty? The incremental character of Swedish climate change policy-making. *Policy Studies* 1-18.
- Konidari, P. and Mavrakakis, D. (2007) A multi-criteria evaluation method for climate change mitigation policy instruments. *Energy Policy* 35, 6235-6257.
- Kreuzer, M. and Hindrichsen, I. K. (2006) Methane mitigation in ruminants by dietary means: The role of their methane emission from manure. *International Congress Series* 1293, 199-208.
- Kronvang, B., Andersen, H. E., Jørgensen, C., Dalgaard, T., Larsen, S. r., Jørgensen, J. and Blicher-Mathiasen, G. (2008) Effects of policy measures implemented in Denmark on nitrogen pollution of the aquatic environment. *Environmental Science & Policy* 11, 144-152.
- Krutilla, K. (2011) Transaction costs and environmental policy: An assessment framework and literature review. *International Review of Environmental & Resource Economics* 4, 261-354.
- Kulling, D. R., Dohme, F., Menzi, H., Sutter, F., Lischer, P. and Kreuzer, M. (2002) Methane emissions of differently fed dairy cows and corresponding methane and nitrogen emissions from their manure during storage. *Environmental Monitoring & Assessment* 79, 129-150.
- Kulling, D. R., Menzi, H., Krober, T. F., Neftel, A., Sutter, F., Licher, P. and Kreuzer, M. (2001) Emissions of ammonia, nitrous oxide and methane from different types of dairy manure during storage as affected by dietary protein content. *The Journal of Agricultural Science* 137, 235-250.
- Lee, C., Hristov, A., Dell, C., Feyereisen, G., Kaye, J. and Beegle, D. (2012) Effect of dietary protein concentration on ammonia and greenhouse gas emitting potential of dairy manure. *J Dairy Sci* 95, 1930-1941.
- Lempert, R. and Schlesinger, M. (2000) Robust Strategies for Abating Climate Change. *Climatic Change* 45, 387-401.
- Lengers, B. and Britz, W. (2012) The choice of emission indicators in environmental policy design: an analysis of GHG abatement in different dairy farms based on a bio-economic



model approach. Review of Agricultural and Environmental Studies - Revue d'Etudes en Agriculture et Environnement 93, 117-144.

Lenzen, M., Moran, D., Kanemoto, K. and Geschke, A. (2013) Building Eora: A global multi-region input-output database at high country and sector resolution. Economic Systems Research 25, 20-49.

Linkov, I., Satterstrom, F. K., Kiker, G., Batchelor, C., Bridges, T. and Ferguson, E. (2006) From comparative risk assessment to multi-criteria decision analysis and adaptive management: Recent developments and applications. Environment International 32, 1072-1093.

Lord, E. I. (1992) Modelling of nitrate leaching: Nitrate Sensitive Areas. Aspects of Applied Biology 30, 19-28.

Lord, E. I. and Anthony, S. G. (2000) MAGPIE: A modelling framework for evaluating nitrate losses at national and catchment scales. Soil Use Manage 16, 167-174.

Louhichi, K., Kanellopoulos, A., Janssen, S., Flichman, G., Blanco, M., Hengsdijk, H., Heckeleei, T., Berentsen, P., Lansink, A. O. and Ittersum, M. v. (2010) FSSIM, a bio-economic farm model for simulating the response of EU farming systems to agricultural and environmental policies. Agricultural Systems 103, 585-597.

Maart-Noelck, S. C. and Musshoff, O. (2013) Investing today or tomorrow? An experimental approach to farmers' decision behaviour. J Agric Econ 64, 295-318.

MacLeod, M., Gerber, P., Mottet, A., Tempio, G., Falcucci, A., Opio, C., Vellinga, T., Henderson, B. and Steinfeld, H. (2013) Greenhouse gas emissions from pig and chicken supply chains - A global life cycle assessment, Food and Agriculture Organization of the United Nations (FAO), Rome.

MacLeod, M., Moran, D., Eory, V., Rees, R. M., Barnes, A., Topp, C. F. E., Ball, B., Hoad, S., Wall, E., McVittie, A., Pajot, G., Matthews, R., Smith, P. and Moxey, A. (2010a) Developing greenhouse gas marginal abatement cost curves for agricultural emissions from crops and soils in the UK. Agricultural Systems 103, 198-209.

MacLeod, M., Moran, D., McVittie, A., Rees, R., Jones, G., Harris, D., Antony, S., Wall, E., Eory, V., Barnes, A., Topp, C. F. E., Ball, B., Hoad, S. and Eory, L. (2010b) Review and update of UK marginal abatement cost curves for agriculture, Report No Report to CCC.

Maia, G. D. N., Day, V., Gates, R. S. and Taraba, J. L. (2012) Ammonia biofiltration and nitrous oxide generation during the start-up of gas-phase compost biofilters. Atmospheric Environment 46, 659-664.

Martin, C., Morgavi, D. and Doreau, M. (2010) Methane mitigation in ruminants: from microbe to the farm scale. Animal 4, 351-365.

Mc Geough, E., O'Kiely, P., Foley, P., Hart, K., Boland, T. and Kenny, D. (2010a) Methane emissions, feed intake, and performance of finishing beef cattle offered maize silages harvested at 4 different stages of maturity. Journal of Animal Science 88, 1479-1491.

Mc Geough, E., O'Kiely, P., Hart, K., Moloney, A., Boland, T. and Kenny, D. (2010b) Methane emissions, feed intake, performance, digestibility, and rumen fermentation of

- finishing beef cattle offered whole-crop wheat silages differing in grain content. *Journal of Animal Science* 88, 2703-2716.
- McCann, L., Colby, B., Easter, K. W., Kasterine, A. and Kuperan, K. V. (2005) Transaction cost measurement for evaluating environmental policies. *Ecological Economics* 52, 527-542.
- Melse, R. W., Ogink, N. W. M. and Rulkens, W. H. (2009) Air treatment techniques for abatement of emissions from intensive livestock production. *The Open Agriculture Journal* 3, 6-12.
- Melse, R. W. and van der Werf, A. W. (2005) Biofiltration for mitigation of methane emission from animal husbandry. *Environ. Sci. Technol.* 39, 5460-5468.
- Mettepenningen, E., Verspecht, A. and Van Huylenbroeck, G. (2009) Measuring private transaction costs of European agri-environmental schemes. *Journal of Environmental Planning and Management* 52, 649-667.
- Meyer-Aurich, A., Schattauer, A., Hellebrand, H. J., Klauss, H., Plochl, M. and Berg, W. (2012) Impact of uncertainties on greenhouse gas mitigation potential of biogas production from agricultural resources. *Renewable Energy* 37, 277-284.
- Misselbrook, T., Chadwick, D., Ghilespy, S. L., Chambers, B., Smith, K. A., Williams, J. and Dragosits, U. (2012) Inventory of ammonia emissions from UK agriculture - 2011, Report No Defra AC0112, Defra, North Wyke Research, ADAS, CEH, London.
- Misselbrook, T. H., Chadwick, D. R., Pain, B. F. and Headon, D. (1998) Dietary manipulation as a means of decreasing N losses and methane emissions and improving herbage N uptake following application of pig slurry to grassland. *The Journal of Agricultural Science* 130, 183-191.
- Misselbrook, T. H., Van der Weerden, T. J., Pain, B. F., Jarvis, S. C., Chambers, B. J., Smith, K. A., Phillips, V. R. and Demmers, T. G. M. (2000) Ammonia emission factors for UK agriculture. *Atmospheric Environment* 34, 871-880.
- Moe, P. W. and Tyrrell, H. F. (1979) Methane production in dairy-cows. *J Dairy Sci* 62, 1583-1586.
- Monteny, G. J., Bannink, A. and Chadwick, D. (2006) Greenhouse gas abatement strategies for animal husbandry. *Agriculture Ecosystems & Environment* 112, 163-170.
- Monteny, G. J. and Erisman, J. W. (1998) Ammonia emission from dairy cow buildings: A review of measurement techniques, influencing factors and possibilities for reduction. *Netherlands Journal of Agricultural Science* 46, 225-247.
- Moorby, J. M., Evans, R. T., Scollan, N. D., Macraet, J. C. and Theodorou, M. K. (2006) Increased concentration of water-soluble carbohydrate in perennial ryegrass (*Lolium perenne* L.). Evaluation in dairy cows in early lactation. *Grass Forage Sci* 61, 52-59.
- Moran, D., MacLeod, M., Wall, E., Eory, V., McVittie, A., Barnes, A., Rees, R., Topp, C. F. E. and Moxey, A. (2011a) Marginal abatement cost curves for UK agricultural greenhouse gas emissions. *J Agric Econ* 62, 93-118.

- Moran, D., MacLeod, M., Wall, E., Eory, V., McVittie, A., Barnes, A., Rees, R. M., Topp, C. F. E., Pajot, G., Matthews, R., Smith, P. and Moxey, A. (2011b) Developing carbon budgets for UK agriculture, land-use, land-use change and forestry out to 2022. *Climatic Change* 105, 529-553.
- Moran, D., MacLeod, M., Wall, E., Eory, V., Pajot, G., Matthews, R., McVittie, A., Barnes, A., Rees, R., Moxey, A., Williams, A. and Smith, P. (2008) UK marginal abatement cost curves for the agriculture and land use, land-use change and forestry sectors out to 2022, with qualitative analysis of options to 2050, Report No RMP4950, Committee on Climate Change, SAC.
- Nahm, K. H. (2002) Efficient feed nutrient utilization to reduce pollutants in poultry and swine manure. *Critical Reviews in Environmental Science and Technology* 32, 1-16.
- Naucler, T. and Enkvist, P. A. (2009) Pathways to Low Carbon Economy: Version 2 of the Global Greenhouse Gas Abatement Cost Curve, McKinsey & Company.
- Nemet, G., Holloway, T. and Meier, P. (2010) Implications of incorporating air-quality co-benefits into climate change policymaking. *Environmental Research Letters* 5, 9.
- Neufeldt, H. and Schafer, M. (2008) Mitigation strategies for greenhouse gas emissions from agriculture using a regional economic-ecosystem model. *Agriculture Ecosystems & Environment* 123, 305-316.
- Nordhaus, W. and Boyer, J. (1999) *Warming the World - Economic Models of Global Warming*, Internet Edition ed. MIT Press.
- O'Neill, D. (2007) The Total External Environmental Costs and Benefits of Agriculture in the UK, EPA.
- OECD (2008) *Handbook on Constructing Composite Indicators*, OECD.
- Oenema, O., Salomez, J., Branquinho, C., Budnakova, M., Cermak, P., Geupel, M., Johnes, P., Tompkins, C., Spranger, T., Erisman, J. W., Palliere, C., Maene, L., Alonso, R., Maas, R., Magid, J., Sutton, M. A. and van Grinsven, H. (2011) Developing integrated approaches to nitrogen management, In: edited by M. A. Sutton, C. M. Howard, J. W. Erisman, G. Billen, A. Bleeker, P. Grennfelt, H. Grinsven, & B. Grizzetti (ed) *The European Nitrogen Assessment*, First Cambridge University Press, Cambridge, pp. 541-550.
- Olander, L., Wollenberg, E., Tubiello, F. and Herold, M. (2013) Advancing agricultural greenhouse gas quantification. *Environmental Research Letters* 8, 011002.
- Oltjen, J. W. and Beckett, J. L. (1996) Role of ruminant livestock in sustainable agricultural systems. *Journal of Animal Science* 74, 1406-1409.
- Opio, C., Gerber, P., Mottet, A., Falcucci, A., Tempio, G., MacLeod, M., Vellinga, T., Henderson, B. and Steinfeld, H. (2013) Greenhouse gas emissions from ruminant supply chains - A global life cycle assessment, Food and Agriculture Organization of the United Nations, Rome.
- Osada, T., Kuroda, K. and Yonaga, M. (2000) Determination of nitrous oxide, methane, and ammonia emissions from a swine waste composting process. *Journal of Material Cycles and Waste Management* 2, 51-56.

- Owens, F. N., Secrist, D. S., Hill, W. J. and Gill, D. R. (1998) Acidosis in cattle: A review. *Journal of Animal Science* 76, 275-286.
- Pannell, D. J., Marshall, G. R., Barr, N., Curtis, A., Vanclay, F. and Wilkinson, R. (2006) Understanding and promoting adoption of conservation practices by rural landholders. *Aust. J. Exp. Agric.* 46, 1407-1424.
- Pape, D., Moffroid, K. and Thompson, V. (2008) Analysis of the potential and costs for greenhouse gas emission reductions within the New Zealand agricultural sector, ICF International, Ministry of Agriculture and Forestry.
- Parkinson, R., Gibbs, P., Burchett, S. and Misselbrook, T. (2004) Effect of turning regime and seasonal weather conditions on nitrogen and phosphorus losses during aerobic composting of cattle manure. *Bioresource Technology* 91, 171-178.
- Pearce, D. W. and Turner, R. K. (1989) Economics of natural resources and the environment. Johns Hopkins University Press Baltimore.
- Pellerin, S., Bamiere, L., Angers, D., Beline, F., Benoit, M., Butault, J. P., Chenu, C., Colnenne-David, C., De Cara, S., Delame, N., Dureau, M., Dupraz, P., Faverdin, P., Garcia-Launay, F., Hassouna, M., Henault, C., Jeuffroy, M. H., Klumpp, K., Metay, A., Moran, D., Recous, S., Samson, E. and Savini, I. (2013) How can French agriculture contribute to reducing greenhouse gas emissions? Abatement potential and cost of ten technical measures, INRA.
- Petersen, S. and Ambus, P. (2006) Methane oxidation in pig and cattle slurry storages, and effects of surface crust moisture and methane availability. *Nutrient Cycling in Agroecosystems* 74, 1-11.
- Petersen, S. O., Amon, B. and Gattinger, A. (2005) Methane oxidation in slurry storage surface crusts. *J. Environ. Qual.* 34, 455-461.
- Petersen, S. O., Andersen, A. J. and Eriksen, J. (2012) Effects of cattle slurry acidification on ammonia and methane evolution during storage. *J. Environ. Qual.* 41, 88-94.
- Petersen, S. O., Dorno, N., Lindholm, S., Feilberg, A. and Eriksen, J. O. (2013) Emissions of CH<sub>4</sub>, N<sub>2</sub>O, NH<sub>3</sub> and odorants from pig slurry during winter and summer storage. *Nutr Cycl Agroecosyst* 95, 103-113.
- Petersen, S. O., Lind, A. M. and Sommer, S. G. (1998) Nitrogen and organic matter losses during storage of cattle and pig manure. *Journal of Agricultural Science* 130, 69-79.
- Petersen, S. O. and Miller, D. N. (2006) Greenhouse gas mitigation by covers on livestock slurry tanks and lagoons? *Journal of the Science of Food and Agriculture* 86, 1407-1411.
- Petersen, S. O., Skov, M., Droscher, P. and Adamsen, A. P. S. (2009) Pilot scale facility to determine gaseous emissions from livestock slurry during storage. *J. Environ. Qual.* 38, 1560-1568.
- Peterson, S. (2006) Uncertainty and economic analysis of climate change: A survey of approaches and findings. *Environmental Modeling & Assessment* 11, 1-17.

Peyraud, J. L. and Astigarraga, L. (1998) Review of the effect of nitrogen fertilization on the chemical composition, intake, digestion and nutritive value of fresh herbage: consequences on animal nutrition and N balance. *Animal Feed Science and Technology* 72, 235-259.

Philippe, F., Laitat, M., Canart, B., Vandenheede, M. and Nicks, B. (2007) Comparison of ammonia and greenhouse gas emissions during the fattening of pigs, kept either on fully slatted floor or on deep litter. *Livestock Science* 111, 144-152.

Philippe, F. X. and Nicks, B. (2015) Review on greenhouse gas emissions from pig houses: Production of carbon dioxide, methane and nitrous oxide by animals and manure. *Agriculture, Ecosystems & Environment* 199, 10-25.

Philippe, F. X., Laitat, M., Canart, B., Farnir, F., Massart, L., Vandenheede, M. and Nicks, B. (2006) Effects of a reduction of diet crude protein content on gaseous emissions from deep-litter pens for fattening pigs. *Animal Research* 55, 397-407.

Pimentel, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Shpritz, L., Fitton, L., Saffouri, R. and Blair, R. (1995) Environmental and Economic Costs of Soil Erosion and Conservation Benefits. *Science* 267, 1117-1123.

Pretty, J. N., Brett, C., Gee, D., Hine, R. E., Mason, C. F., Morison, J. I. L., Raven, H., Rayment, M. D. and van der Bijl, G. (2000) An assessment of the total external costs of UK agriculture. *Agricultural Systems* 65, 113-136.

Pretty, J. N., Mason, C. F., Nedwell, D. B. and Hine, R. E. (2002) A Preliminary Assessment of the Environmental Costs of the Eutrophication of Fresh Waters in England and Wales, University of Essex, Colchester UK.

Pretty, J., Brett, C., Gee, D., Hine, R., Mason, C., Morison, J., Rayment, M., Van Der Bijl, G. and Dobbs, T. (2001) Policy Challenges and Priorities for Internalizing the Externalities of Modern Agriculture. *Journal of Environmental Planning and Management* 44, 263-283.

Price, R., Thornton, S. and Nelson, S. (2007) The social cost of carbon and the shadow price of carbon: What they are, and how to use them in economic appraisal in the UK, Department for Environment, Food and Rural Affairs; Crown copyright, London.

Ravishankara, A. R., Daniel, J. S. and Portmann, R. W. (2009) Nitrous oxide (N<sub>2</sub>O): The dominant ozone-depleting substance emitted in the 21<sup>st</sup> century. *Science* 326, 123-125.

Reilly, J., Prinn, R., Harnisch, J., Fitzmaurice, J., Jacoby, H., Kicklighter, D., Melillo, J., Stone, P., Sokolov, A. and Wang, C. (1999) Multi-gas assessment of the Kyoto Protocol. *Nature* 401, 549-555.

Reis, S., Nitter, S. and Friedrich, R. (2005) Innovative approaches in integrated assessment modelling of European air pollution control strategies – Implications of dealing with multi-pollutant multi-effect problems. *Environmental Modelling & Software* 20, 1524-1531.

Rennenberg, H., Dannenmann, M., Gessler, A., Kreuzwieser, J., Simon, J. and Papen, H. (2009) Nitrogen balance in forest soils: nutritional limitation of plants under climate change stresses. *Plant Biology* 11, 4-23.

- Renwick, A., Jansson, T., Verburg, P. H., Revoredo-Giha, C., Britz, W., Gocht, A. and McCracken, D. (2013) Policy reform and agricultural land abandonment in the EU. *Land Use Policy* 30, 446-457.
- Roberts, D. J., Bell, J., Bevan, K. and McEvoy, T. (2008) Beyond calf exports: the efficacy, economics and practicalities of sexed semen as a welfare-friendly herd replacement tool in the dairy industry, Report No Report produced for Compassion in World Farming & The Royal Society for the Prevention of Cruelty to Animals, SAC.
- Roche, J. F. (2006) The effect of nutritional management of the dairy cow on reproductive efficiency. *Animal Reproduction Science* 96, 282-296.
- Rorstad, P. K., Vatn, A. and Kvakkestad, V. (2007) Why do transaction costs of agricultural policies vary? *Agricultural Economics* 36, 1-11.
- Rotz, C. A. (2004) Management to reduce nitrogen losses in animal production. *Journal of Animal Science* 82, E119-E137.
- SAC (2012) The Farm Management Handbook 2012/13, SAC Consulting, Rural Business Unit, Bush Estate, SRUC.
- Sanz-Cobena, A., Sanchez-Martin, L., Garcia-Torres, L. and Vallejo, A. (2012) Gaseous emissions of N<sub>2</sub>O and NO and NO<sub>3</sub><sup>-</sup> leaching from urea applied with urease and nitrification inhibitors to a maize (*Zea mays*) crop. *Agriculture Ecosystems & Environment* 149, 64-73.
- Schils, R. L. M., Olesen, J. E., del Prado, A. and Soussana, J. F. (2007) A review of farm level modelling approaches for mitigating greenhouse gas emissions from ruminant livestock systems. *Livestock Science* 112, 240-251.
- Schneider, U. A., Mccarl, B. A. and Schmid, E. (2007) Agricultural sector analysis on greenhouse gas mitigation in US agriculture and forestry. *Agricultural Systems* 94, 128-140.
- Scholefield, D. and Haygarth, P. (2004) Cost curve of nitrate mitigation options, Report No Defra NT2511, Defra, IGER, London.
- Schulte, R. P., Crosson, P., Donnellan, T., Farrelly, N., Finnan, J., Lalor, S. T., Lanigan, G., O'Brien, D., Shalloo, L. and Thorne, F. Schulte, R. P. and Donnellan, T. (ed) (2012) A marginal abatement cost curve for Irish agriculture, Teagasc.
- Scottish Executive Reid, A. (ed) (2011) Economic report on Scottish agriculture - 2011 edition, Scottish Executive Environment and Rural Affairs Department, Economics and Statistics, Edinburgh.
- Scottish Government (2011a) Low Carbon Scotland: Meeting the Emissions Reduction Targets 2010-2022 - The Report on Proposals and Policies, Scottish Government, Edinburgh.
- Scottish Government (2011b) Low Carbon Scotland: Meeting the Emissions Reduction Targets 2010-2022 Technical Appendix - The Report on Proposals and Policies, Scottish Government, Edinburgh.

Scottish Government (2013a) Low Carbon Scotland: Meeting the Emissions Reduction Targets 2010-2022 - The Second Report on Proposals and Policies, Scottish Government, Edinburgh.

Scottish Government (2013b) Scottish Greenhouse Gas Emissions 2011, Scottish Government.

Scottish Government (2014) United Kingdom - Rural Development Programme (Regional) - Scotland, Report No CCI: 2014UK06RDRP003.

Shepherd, M. A., Anthony, S., Temple, M., Burgess, D., Patton, M., Renwick, A., Barnes, A. and Chadwick, D. (2007) Baseline Projections for Agriculture and implications for emissions to air and water, Report No Defra SFF0601, Defra, ADAS, SAC, IGER, London.

Shrestha, S. (2004). Adaptation strategies for dairy farms in central and north-west England under climate change, UK, University of Nottingham.

Shrestha, S., Ciaian, P., Himics, M. and Van Doorslaer, B. (2013) Impacts of climate change on EU agriculture. Review of Agricultural and Applied Economics 16, 24-39.

Shrestha, S. and Hennessy, T. (2006) Changes in land use on farms in the Border and the Mid-East regions of Ireland after decoupling of farm payments. Agricultural Research Forum, 15-16 March, 2006, Tullamore, Ireland,

Shrestha, S., Hennessy, T. and Hynes, S. (2007) The effect of decoupling on farming in Ireland: A regional analysis. Irish Journal of Agriculture and Food Research 46, 1-13.

Silsoe Research Institute (2000) The effects of covering slurry stores on emissions of ammonia, methane and nitrous oxide, Report No Defra WA0625, Defra, London.

Smart, J. C., Hicks, K., Morrissey, T., Heinemeyer, A., Sutton, M. A. and Ashmore, M. (2011) Applying the ecosystem service concept to air quality management in the UK: a case study for ammonia. Environmetrics 22, 649-661.

Smil, V. (1999) Detonator of the population explosion. Nature 400, 415.

Smith, L. A. and Stern, N. (2011) Uncertainty in science and its role in climate policy. Philosophical Transactions of the Royal Society A-Mathematical Physical and Engineering Sciences 369, 4818-4841.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B. and Sirotenko, O. (2007) Agriculture, In: edited by B. Metz, O. R. Davidson, P. R. Bosch, R. Dave, & L. A. Meyer (ed) Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA

Smith, P. (2011) Agricultural greenhouse gas mitigation potential globally, in Europe and in the UK: what have we learnt in the last 20-years? Glob. Change Biol. n/a.

Snyder, C. S., Bruulsema, T. W., Jensen, T. L. and Fixen, P. E. (2009) Review of greenhouse gas emissions from crop production systems and fertilizer management effects. Agriculture Ecosystems & Environment 133, 247-266.

- Soderqvist, T. and Hasselstrom, L. (2008) The economic value of ecosystem services provided by the Baltic Sea and Skagerrak, Report No Swedish Environmental Protection Agency 5874, Swedish Environmental Protection Agency.
- Solano, C., Leon, H., Perez, E. and Herrero, M. (2001) Characterising objective profiles of Costa Rican dairy farmers. *Agricultural Systems* 67, 153-179.
- Sommer, S. G. (2001) Effect of composting on nutrient loss and nitrogen availability of cattle deep litter. *European Journal of Agronomy* 14, 123-133.
- Sommer, S. G. and Moller, H. B. (2000) Emission of greenhouse gases during composting of deep litter from pig production - effect of straw content. *Journal of Agricultural Science* 134, 327-335.
- Sommer, S. G., Olesen, J. E. and Christensen, B. T. (1991) Effects of temperature, wind speed and air humidity on ammonia volatilization from surface applied cattle slurry. *The Journal of Agricultural Science* 117, 91-100.
- Sommer, S. G., Petersen, S. O. and Moller, H. B. (2004) Algorithms for calculating methane and nitrous oxide emissions from manure management. *Nutrient Cycling in Agroecosystems* 69, 143-154.
- Sommer, S. G., Petersen, S. O. and Sogaard, H. T. (2000) Greenhouse Gas Emission from Stored Livestock Slurry. *J. Environ. Qual.* 29, 744-751.
- Sommer, S. G., Olesen, J. E., Petersen, S. O., Weisbjerg, M. R., Valli, L., Rodhe, L. and Beline, F. (2009) Region-specific assessment of greenhouse gas mitigation with different manure management strategies in four agroecological zones. *Glob. Change Biol.* 15, 2825-2837.
- Spencer, I., Bann, C., Moran, D., McVittie, A., Lawrence, K. and Caldwell, V. (2008) Environmental Accounts for Agriculture, Report No Defra SFS0601, Defra, Welsh Assembly Government, Scottish Government, Department of Agriculture & Rural Development (N. Ireland), SAC, Cranfield University.
- St-Pierre, N. R. and Thraen, C. S. (1999) Animal grouping strategies, sources of variation, and economic factors affecting nutrient balance on dairy farms. *Journal of Animal Science* 77, 72-83.
- Stehfest, E. and Bouwman, L. (2006) N<sub>2</sub>O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modeling of global annual emissions. *Nutr Cycl Agroecosyst* 74, 207-228.
- Szanto, G., Hamelers, H., Rulkens, W. and Veeken, A. (2007) NH<sub>3</sub>, N<sub>2</sub>O and CH<sub>4</sub> emissions during passively aerated composting of straw-rich pig manure. *Bioresource Technology* 98, 2659-2670.
- Thistlethwait, G., Goodwin, J., Salisbury, E., MacCarthy, J., Pang, Y., Thomson, A. and Cardenas, L. (2012) Greenhouse Gas Inventories for England, Scotland, Wales and Northern Ireland: 1990 - 2010, Report No Width1AEA/ENV/R/3314 Issue 1 Width1, Width156968/2010/CD7683/JMC Width1, AEA.



Thomas, J., Thistlethwait, G., MacCarthy, J., Pearson, B., Murrels, T., Pang, Y., Passant, N., Webb, N., Connolly, C., Cardenas, L., Malcolm, H. and Thomson, A. (2011) Greenhouse Gas Inventories for England, Scotland, Wales and Northern Ireland: 1990 - 2009, Report No AEAT/ENV/R/3222 Issue 1, ED56403200/Issue 1, AEA.

Thorman, R., Webb, J., Yamulki, S., Chadwick, D. R., Bennett, G., McMillan, S., Kingston, H., Donovan, N. and Misselbrook, T. (2008) The effect of solid manure incorporation on nitrous oxide emissions. 13th International Conference of the FAO RAMIRAN: Potential for Simple Technology Solutions in Organic Manure Management. Ambrozia NT Ltd., Albena, Bulgaria, Proceedings of the 13th International Conference of the FAO RAMIRAN: Potential for Simple Technology Solutions in Organic Manure Management. Ambrozia NT Ltd., Albena, Bulgaria,

Tol, R. (2012) On the uncertainty about the total economic impact of climate change. *Environmental and Resource Economics* 53, 97-116.

Tol, R. S. J. (1999) Safe policies in an uncertain climate: an application of FUND. *Global Environmental Change* 9, 221-232.

Tol, R. S. J. (2005) The marginal damage costs of carbon dioxide emissions: an assessment of the uncertainties. *Energy Policy* 33, 2064-2074.

van Grinsven, H. J., Rabl, A. and de Kok, T. M. (2010) Estimation of incidence and social cost of colon cancer due to nitrate in drinking water in the EU: a tentative cost-benefit assessment. *Environmental Health* 9, 58.

van Vuuren, D. P., Bouwman, L. F., Smith, S. J. and Dentener, F. (2011) Global projections for anthropogenic reactive nitrogen emissions to the atmosphere: an assessment of scenarios in the scientific literature. *Current Opinion in Environmental Sustainability* 3, 359-369.

van Vuuren, D. P., Weyant, J. and de la Chesnaye, F. (2006) Multi-gas scenarios to stabilize radiative forcing. *Energy Economics* 28, 102-120.

VandeHaar, M. J. and St-Pierre, N. (2006) Major advances in nutrition: Relevance to the sustainability of the dairy industry. *J Dairy Sci* 89, 1280-1291.

VanderZaag, A. C., Gordon, R. J., Glass, V. M. and Jamieson, R. C. (2008) Floating covers to reduce gas emissions from liquid manure storages: A review. *Applied Engineering in Agriculture* 24, 657-671.

VanderZaag, A. C., Gordon, R. J., Jamieson, R. C., Burton, D. L. and Stratton, G. W. (2009) Gas emissions from straw covered liquid dairy manure during summer storage and autumn agitation. *Transactions of the Asabe* 52, 599-608.

VanderZaag, A. C., Gordon, R. J., Jamieson, R. C., Burton, D. L. and Stratton, G. W. (2010) Permeable synthetic covers for controlling emissions from liquid dairy manure. *Applied Engineering in Agriculture* 26, 287-297.

Vellinga, T. V. and Hoving, I. (2011) Maize silage for dairy cows: mitigation of methane emissions can be offset by land use change. *Nutrient Cycling in Agroecosystems* 89, 413-426.

- Velthof, G. L., Nelemans, J. A., Oenema, O. and Kuikman, P. J. (2005) Gaseous nitrogen and carbon losses from pig manure derived from different diets. *J. Environ. Qual.* 34, 698-706.
- Vermeulen, S. J., Challinor, A. J., Thornton, P. K., Campbell, B. M., Eriyagama, N., Vervoort, J. M., Kinyangi, J., Jarvis, A., Laderach, P., Ramirez-Villegas, J., Nicklin, K. J., Hawkins, E. and Smith, D. R. (2013) Addressing uncertainty in adaptation planning for agriculture. *Proceedings of the National Academy of Sciences* 110, 8357-8362.
- Vermont, B. and De Cara, S. (2010) How costly is mitigation of non-CO<sub>2</sub> greenhouse gas emissions from agriculture?: A meta-analysis. *Ecological Economics* 69, 1373-1386.
- Vistoso, E., Alfaro, M., Saggar, S. and Salazar, F. (2012) Effect of nitrogen inhibitors on nitrous oxide emissions and pasture growth after an autumn application in volcanic soil. *Chilean Journal of Agricultural Research* 72, 133-139.
- Vogt-Schilb, A. and Hallegatte, S. (2011) When Starting with the Most Expensive Option Makes Sense - Use and Misuse of Marginal Abatement Cost Curves, Report No Policy Research Working Paper 5803, The World Bank, Sustainable Development Network, Office of the Chief Economist.
- Wagner, F., Amann, M., Borken-Kleefeld, J., Cofala, J., Hoglund-Isaksson, L., Purohit, P., Rafaj, P., Schopp, W. and Winiwarter, W. (2012) Sectoral marginal abatement cost curves: implications for mitigation pledges and air pollution co-benefits for Annex I countries. *Sustain Sci* 7, 169-184.
- Waldrip, H. M., Todd, R. W. and Cole, N. A. (2013) Prediction of nitrogen excretion by beef cattle: A meta-analysis. *Journal of Animal Science* 91, 4290-4302.
- Webb, J. (2001) Estimating the potential for ammonia emissions from livestock excreta and manures. *Environmental Pollution* 111, 395-406.
- Webb, J. and Misselbrook, T. H. (2004) A mass-flow model of ammonia emissions from UK livestock production. *Atmospheric Environment* 38, 2163-2176.
- Webb, J., Pain, B., Bittman, S. and Morgan, J. (2010) The impacts of manure application methods on emissions of ammonia, nitrous oxide and on crop response - A review. *Agriculture Ecosystems & Environment* 137, 39-46.
- Webb, J., Ryan, M., Anthony, S. G., Brewer, A., Laws, J., Aller, M. F. and Misselbrook, T. H. (2006) Cost-effective means of reducing ammonia emissions from UK agriculture using the NARSES model. *Atmospheric Environment* 40, 7222-7233.
- Weiske, A. (2005) Survey of technical and management-based mitigation measures in agriculture, Report No MEACAP WP3 D7a, Institute for Energy and Environment (IE).
- Westhoek, H., Lesschen, J. P., Rood, T., Wagner, S., De Marco, A., Murphy-Bokern, D., Leip, A., van Grinsven, H., Sutton, M. A. and Oenema, O. (2014) Food choices, health and environment: Effects of cutting Europe's meat and dairy intake. *Global Environmental Change*.

Winiwarter, W. and Klimont, Z. (2011) The role of N-gases ( $\text{N}_2\text{O}$ ,  $\text{NO}_x$ ,  $\text{NH}_3$ ) in cost-effective strategies to reduce greenhouse gas emissions and air pollution in Europe. *Current Opinion in Environmental Sustainability* 3, 438-445.

Yamulki, S. (2006) Effect of straw addition on nitrous oxide and methane emissions from stored farmyard manures. *Agriculture Ecosystems & Environment* 112, 140-145.

Zaman, M., Saggar, S., Blennerhassett, J. D. and Singh, J. (2009) Effect of urease and nitrification inhibitors on N transformation, gaseous emissions of ammonia and nitrous oxide, pasture yield and N uptake in grazed pasture system. *Soil Biology and Biochemistry* 41, 1270-1280.

Zhang, G., Strom, J. S., Li, B., Rom, H. B., Morsing, S., Dahl, P. and Wang, C. (2005) Emission of ammonia and other contaminant gases from naturally ventilated dairy cattle buildings. *Biosystems Engineering* 92, 355-364.

## Appendix – Papers published

Available online at [www.sciencedirect.com](http://www.sciencedirect.com)

SciVerse ScienceDirect

journal homepage: [www.elsevier.com/locate/envsci](http://www.elsevier.com/locate/envsci)

# Multiple-pollutant cost-effectiveness of greenhouse gas mitigation measures in the UK agriculture

Vera Eory\*, Cairistiona F.E. Topp, Dominic Moran

Research Division, SRUC, West Mains Road, Edinburgh EH9 3JG, UK

## ARTICLE INFO

### Article history:

Received 1 August 2012

Received in revised form

9 November 2012

Accepted 9 November 2012

Published on line

### Keywords:

Marginal abatement costs curves

Cost-effectiveness

Greenhouse gases

Co-effects

Nitrogen

Phosphorus

## ABSTRACT

This paper develops multiple-pollutant marginal abatement cost curve analysis to identify an optimal set of greenhouse gas (GHG) mitigation measures considering the trade-offs and synergies with other environmental pollutants. The analysis is applied to UK agriculture, a sector expected to make a contribution to the national GHG mitigation effort. Previous analyses using marginal abatement cost curves (MACCs) have determined the sector's GHG abatement potential based on the cost-effectiveness of a variety of technically feasible mitigation measures. Most of these measures have external effects on other pollution loads arising from agricultural activities. Here the monetary values of four of the most important impacts to water and air (specifically ammonia, nitrate, phosphorous and sediment) are included in the cost-effectiveness analysis. The resulting multiple-pollutant marginal abatement cost curve (MP MACC) informs the design of sustainable climate change policies by showing how the MP MACC for the UK agriculture can differ from the GHG MACC. The analysis also highlights research gaps, and suggests a need to understand the wider environmental effects of GHG mitigation options and to reduce the uncertainty in pollutant damage cost estimates.

© 2012 Elsevier Ltd. All rights reserved.

## 1. Introduction

Climate change mitigation is high on the environmental policy agenda as countries seek to meet emissions reduction commitments. Agriculture is an important source of GHG emissions, accounting for 10–12% of total global and 9% of UK GHG emissions (Smith et al., 2007; Thomas et al., 2011). The sector is thought to offer significant emission reduction potential through the deployment of a number of cost-effective mitigation and carbon sequestration measures. But the implementation of these measures can occasion other environmental impacts that need to be addressed in any overall assessment of measure cost-effectiveness.

Land based mitigation measures can be highly variable in terms of their emission reduction (abatement) potential and private cost of measure implementation. Moreover, some

measures have wider environmental co-effects (external effects), that can be both positive and negative. Adding these co-effects to the private cost of measures defines a social cost that can be used to redefine the cost-effectiveness of measures (i.e. the costs of implementation relative to GHG benefits). This paper investigates the social cost of GHG mitigation measures and aims to outline a more accurate cost-effectiveness metric for ranking measures in a marginal abatement cost curve.

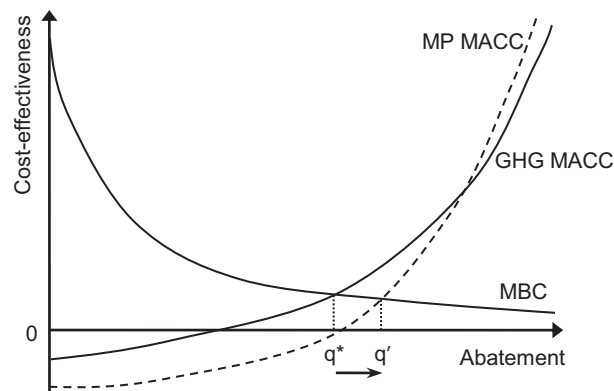
Marginal abatement cost curves (MACCs) are tools to identify relatively cost-effective mitigation measures (MMs) across the economy (Kesicki and Strachan, 2011). MACCs can also be used to define the economically optimal level of abatement, where marginal abatement costs are equal to the resulting marginal benefits (Pearce and Turner, 1990). In practice, the economically optimal level of GHG abatement is defined by comparing marginal abatement costs with a standard benefit benchmark such as the shadow price of carbon.

\* Corresponding author. Tel.: +44 131 535 4313; fax: +44 131 535 4345.

E-mail address: [vera.eory@sruc.ac.uk](mailto:vera.eory@sruc.ac.uk) (V. Eory).

1462-9011/\$ – see front matter © 2012 Elsevier Ltd. All rights reserved.

<http://dx.doi.org/10.1016/j.envsci.2012.11.003>



**Fig. 1 – Effect of inclusion of co-effects on the GHG MAC curve and on the economic optimum of pollution control. MBC: marginal benefit curve.  $q^*$ ,  $q'$ : economic optimum of pollution reduction without and with co-effects, respectively.**

Fig. 1 shows how adding external effects can alter the theoretical MACC. Positive co-effects reduce abatement costs, while negative ones increase the abatement cost, thus tilting the curve. The intercept of the MACC and the marginal benefit curve (MBC) indicates the economically optimal level of abatement ( $q^*$ ). In case where the co-effects are mostly positive, the reduced abatement costs result in an increased abatement optimum ( $q'$ ), or in decreased overall costs of achieving a targeted pollution reduction level.

GHG MACCs have been constructed for various sectors including energy and transport (Enkvist et al., 2007), and have galvanised wider debate and action on mitigation policy. MACCs have also been used to inform policy development on measures targeting various agricultural pollutants (see e.g. Webb et al., 2006 for ammonia, Haygarth et al., 2009 for phosphorous and Scholefield and Haygarth, 2004 for nitrates). But these studies have been limited in their treatment of any co-effects and hence the trade-offs and synergies between different agricultural pollutants (Reis et al., 2005).

There is a growing literature modelling multiple pollutants. Brink et al. (2001, 2005) analysed the co-effects of  $\text{NH}_3$  and GHG mitigation options in European agriculture. Wagner et al. (2012) presented a multi-sector assessment of GHG mitigation options and their air pollution co-effects ( $\text{SO}_2$ ,  $\text{NO}_x$ ,  $\text{PM}_{2.5}$ ) in Annex I countries to United Nations Framework Convention on Climate Change. Anthony et al. (2008) provided a cost-benefit assessment of six agricultural pollutants (nitrate, phosphorous, sediment, ammonia, methane and nitrous oxide) for the UK. In the US, Schneider et al. (2007) estimated the external effects of GHG mitigation options on soil erosion, N and P pollution. The optimisation approach in these studies is either based on a single pollutant, or provides the least-cost solution based on specified pollution reduction targets.

In contrast, a MACC can potentially facilitate the representation of the socially optimal abatement potential by accommodating multiple pollutants into a marginal cost curve. This single metric can be generated by monetising environmental co-effects, creating a multiple-pollutant (MP) MACC. Relative to a GHG MACC, an MP MACC also enables better representation of the social cost of integrated policies.

This paper considers the consequences of including available data on the monetary valuation of GHG mitigation

measures' co-effects into the existing GHG MACC estimates developed for agriculture in the UK (Moran et al., 2011b). The external effects included are nitrate leaching, ammonia emissions, phosphorous and sediment pollution. We are unaware of any studies adding co-effects of mitigation effort to MACCs using a single metric of cost-effectiveness.

The rest of the paper is structured as follows. Section two provides more background to the MP MACC analysis in agriculture. Sections three and four outline a methodology for the paper and present results. Sections five and six provide a discussion and a conclusion, respectively.

## 2. Background

Agriculture is expected to make a contribution to the national mitigation effort in the UK that is being coordinated by the UK Committee on Climate Change and partly informed by sector-wide MACC analyses. Technically feasible measures for mitigating GHG emissions in the UK agriculture include, for example, improved resource use efficiency at farm level, generating greater output per unit of input. Higher efficiency can be achieved via selective breeding of livestock, optimised feeding strategies and judicious use of nitrogen fertilisers. Other MMs include changes in animal housing and manure storage, enhancing the removal of atmospheric  $\text{CO}_2$  via sequestration into soil and vegetation sinks and replacing fossil fuel emissions with alternative energy sources.

Earlier GHG MACC analysis identified a financially feasible subset of measures, based on the private costs of implementation and on the abatement potential of the measures (Moran et al., 2011a). The analysis noted the particular biophysical complexities of agricultural mitigation and the likelihood of potentially large co-effects associated with the widespread implementation of many measures. These co-effects could include reduced (or increased) pollution to water, mitigation of other pollutants including ammonia, and more complex impacts to ecosystems functions.

Specific effects considered in this analysis are nitrate leaching, ammonia emissions, phosphorous and sediment pollution. These pollutants are drivers of environmental changes, leading to changes in ecosystem services. Nitrate

and phosphorous cause eutrophication in aquatic ecosystems, and drinking water nitrate levels are controlled to eliminate the risk of methaemoglobinaemia. Sediment in water-bodies (originating from soil erosion) has negative effects on biological water quality, contributes to drinking water contamination, and when deposited by fluvial flooding, can damage property, roads and transport links. Water-borne sediment is only part of the problem arising from soil erosion; other effects of soil erosion are not included in this assessment. Ammonia emissions are associated with human health and environmental issues, most importantly respiratory problems (via the formation of secondary aerosols contributing to particulate matter concentrations above critical levels), acidification and eutrophication (both aquatic and terrestrial). For reviews on processes related to nitrate and ammonia see, for example Chapter 22 and 23 of The European Nitrogen Assessment (Brink et al., 2011; Oenema et al., 2011, respectively), for phosphorous Correll (1998), and for sediment, Pimentel et al. (1995).

Accordingly, the current study draws on existing evidence on both the biophysical impact of other pollutants and available damage costs. Damage costs are monetary estimates of the damage a pollutant causes to society. Here these effects are quantified in monetary terms using evidence from existing non-market valuation literature that can be transferred for use in the MP MACC.

### 3. Methods

#### 3.1. Calculations

In this paper we further develop the GHG MACC elaborated in Moran et al. (2008), where the MACC analysis ranks the mitigation options in decreasing order of cost-effectiveness by dividing the cost of the measure with the GHG abatement achievable. The measures are additional to mitigation activity that would be expected to happen in a 'business as usual' baseline. That analysis was revisited to represent uncertainty of effectiveness assumptions and to further develop existing interaction factors between the measures to avoid double-counting the abatement potential of individual measures (MacLeod et al., 2010b). In that paper alternative empirical estimates were used to approximate uncertainties, leading to the construction of optimistic and pessimistic abatement scenarios based on upper and lower abatement rates, applicability rates and cost estimates for specific mitigation measures. A maximum technical potential (MTP) refers to the level of abatement reached assuming the full potential of all MMs, given full uptake by farmers. In contrast, a lower feasible potential allows for the likelihood of behavioural constraints that suggest that no MM is likely to be adopted to 100%. We use the 2022 MTP Optimistic MACC as a basis for the current analysis.

The cost-effectiveness (CE) calculation underlying the GHG MACC is initially altered to accommodate the additional net external effects associated with each mitigation measure. For the MP MACC the quantitative emission reduction of the MM on each pollutant are multiplied with the damage costs of the pollutants to derive an estimate of the monetary value of the external effects (Eq. (1)). The external effects are then added to

the private cost of the option, providing the social cost of the MM, which is used in the social cost-effectiveness calculation (Eq. (2)). From this point on, the calculation follows the method described in (Moran et al., 2008) and (MacLeod et al., 2010b).

$$\text{external cost}_i = \sum_{j=1}^k \text{change in pollution load}_{i,j} \times \text{damage cost}_j \quad (1)$$

$$\text{social CE}_i = \frac{\text{private cost}_i + \text{external cost}_i}{\text{GHG saved}_i} \quad (2)$$

where  $\text{external cost}_i$  is the monetary value of the external effects of mitigation measure  $i$ ;  $\text{social CE}_i$  is the cost-effectiveness of mitigation measure  $i$  with external effects;  $i$  refers to mitigation measure  $i$  and  $j$  refers to pollutant  $j$ .

#### 3.2. Data sources

MacLeod et al. (2010b) provided data on the range of mitigation measures, their GHG abatement rates and private costs along with interaction factors. In that paper, baseline activity data (animal numbers, land areas) represent 2022 forecasts derived from Shepherd et al. (2007). Applicability and GHG abatement rates of the measures are based on a literature review and expert opinion (MacLeod et al., 2010b). Cost data for crop and soil measures are derived from a representative farm optimisation model (see more in MacLeod et al., 2010a), while livestock costs are adopted from IGER (2001).

The quantity of associated co-effects of measures on nitrate, ammonia, phosphorous and sediment pollution were reported in Anthony et al. (2008), derived from the application of a range of process models: NARSES (Webb and Misselbrook, 2004), NT26AE (Chadwick et al., 2005), MANNER (Chambers et al., 1999), PSYCHIC (Davison et al., 2008), NEAP-N (Lord and Anthony, 2000), NITCAT (Lord, 1992). In that paper the estimated pollution loads were based on activity data for 2004 (Shepherd et al., 2007), the pollution saving values reported account for interactions between the measures to avoid double counting of the pollution reduction. The different base years of the two datasets (Anthony et al., 2008; MacLeod et al., 2010b) lead to a slight discrepancy in the current analysis, resulting in approximately 5% underestimation of external effects for measures applicable to arable land and similar overestimation of co-effects for livestock measures. Note also that the Anthony et al. (2008) study only covered pollution loading in England and Wales and we therefore restrict the subsequent analysis to England and Wales.

The attribution of external effects to mitigation measures required a comparison of the specific MMs being evaluated in the two key studies (Anthony et al., 2008; MacLeod et al., 2010b). A match between the measures was achieved by either amalgamating or disaggregating some measures in one or other of the studies. In some cases no match was possible, resulting in data gaps for some of the MMs on the MP MACC (see Appendix).

Monetary estimates are required to value the emission reduction of each pollutant associated with the measures. To reflect the uncertainty in the damage costs the current analysis uses five sets of damage costs derived from the literature and recent policy reports, covering the four pollutants: nitrate, phosphorous, sediment, ammonia (see

**Table 1 – Unit damage cost sets used in this study.**

Damage cost set	NO <sub>3</sub> -N (£/t)	P (£/t)	Sediment (£/t)	NH <sub>3</sub> -N (£/t)	Resulting MACC
None	–	–	–	–	GHG-MACC
A	217 <sup>a</sup>	9634 <sup>a</sup>	25 <sup>a</sup>	1804 <sup>b</sup>	MP-MACC-A
B	672 <sup>c</sup>	45,144 <sup>c</sup>	108 <sup>c</sup>	1804 <sup>b</sup>	MP-MACC-B
C	4287 <sup>d</sup>	9634 <sup>a</sup>	25 <sup>a</sup>	17,699 <sup>e</sup>	MP-MACC-C
D	4287 <sup>d</sup>	45,144 <sup>c</sup>	108 <sup>c</sup>	17,699 <sup>e</sup>	MP-MACC-D
E	20,577 <sup>d</sup>	45,144 <sup>c</sup>	108 <sup>c</sup>	52,055 <sup>e</sup>	MP-MACC-E

<sup>a</sup> Values derived by (Anthony et al., 2008) after (Spencer et al., 2008).

<sup>b</sup> Defra damage cost (Defra, 2008) as used in (Anthony et al., 2008).

<sup>c</sup> Values derived by (Anthony et al., 2008), based on (Spencer et al., 2008) and (Baker et al., 2007).

<sup>d</sup> Values based on (Brink et al., 2011).

<sup>e</sup> Values based on (Holland et al., 2005).

Table 1). These estimates were applied to the 2022 MTP Optimistic GHG MACC. Sets A and B are described by Anthony et al. (2008), while sets C–E were added to the analysis to represent the higher end of damage cost estimates found in the literature. The 2022 MTP Optimistic GHG MACC is considered in the analysis for comparison.

The sources for the selected damage costs for each external effect are shown in Table 2, which also reflects a variety of methodological approaches to derive monetary values. We note that this form of non-market valuation is problematic both in terms of the differences in methodological approach and the paucity of studies for specific co-effects relating to the pressures under consideration. In general, this means that

some form of benefits transfer is necessary (see Brouwer, 2000), which inevitably introduces some subjectivity into how well existing studies match the external effects under investigation. The carbon price benchmark used in this paper is the shadow price of carbon (SPC), estimated to be 34.3 £ tCO<sub>2</sub>e<sup>−1</sup> in 2022 (Price et al., 2007).

#### 4. Results

Results are first presented on the value of co-effects for each measure where data are available. The analysis will then demonstrate how these benefits affect the shape of the GHG

**Table 2 – Basis of valuation of damage costs.**

Pollutant	Damage cost set	Impact	Basis of valuation	References
NO <sub>3</sub> <sup>−</sup> , P, sed.	NO <sub>3</sub> <sup>−</sup> : A/B P, sed.: A/B/C/D/E	Nitrates in drinking water	Expenditure on nitrate removal by water companies	Spencer et al. (2008)
		Particulates in drinking water	Expenditure on sediment removal by water companies	Spencer et al. (2008)
		Agricultural pollution incidents	Costs of restocking rivers with fish following a pollution incident	Spencer et al. (2008)
		Degraded river quality	Benefit transfer of WTP for water quality of the River Tame	Spencer et al. (2008), Georgiou et al. (2000)
		Degraded estuary quality	Benefit transfer of WTP for water quality of the River Tame	Spencer et al. (2008), Georgiou et al. (2000)
		Lake eutrophication	Reduced value of waterside property, reduced value of water for commercial use, drinking water treatment costs (algae), clean-up costs of waterways, increased GHG and acidifying gases emissions, reduced recreational amenity value, overall ecological damage	Spencer et al. (2008), Pretty et al. (2002), O'Neill (2007)
		Offsite soil erosion damage	Costs of dredging waterways, damage to property and roads	Spencer et al. (2008)
NO <sub>3</sub> <sup>−</sup>	C/D/E	Degraded marine quality	WTP for clean Baltic Sea. Higher and lower bound	Brink et al. (2011), Soderqvist and Hasselstrom (2008), Gren et al. (2008)
		Nitrates in drinking water	Mortality from colon cancer. Higher and lower bound	Brink et al. (2011), van Grinsven et al. (2010)
NH <sub>3</sub>	A/B	Air pollution: SIA	Chronic mortality caused by PM <sub>2.5</sub> .	Defra (2008)
NH <sub>3</sub>	C/D/E	Air pollution: SIA	Chronic mortality, chronic and acute morbidity caused by PM <sub>2.5</sub> , along with effects on crops by hindering tropospheric ozone formation	Holland et al. (2005)

WTP, willingness to pay; SIA, secondary inorganic aerosols; PM<sub>2.5</sub>, fine particles; sed., sediment.



marginal abatement cost curves and hence the overall cost-effective abatement potential.

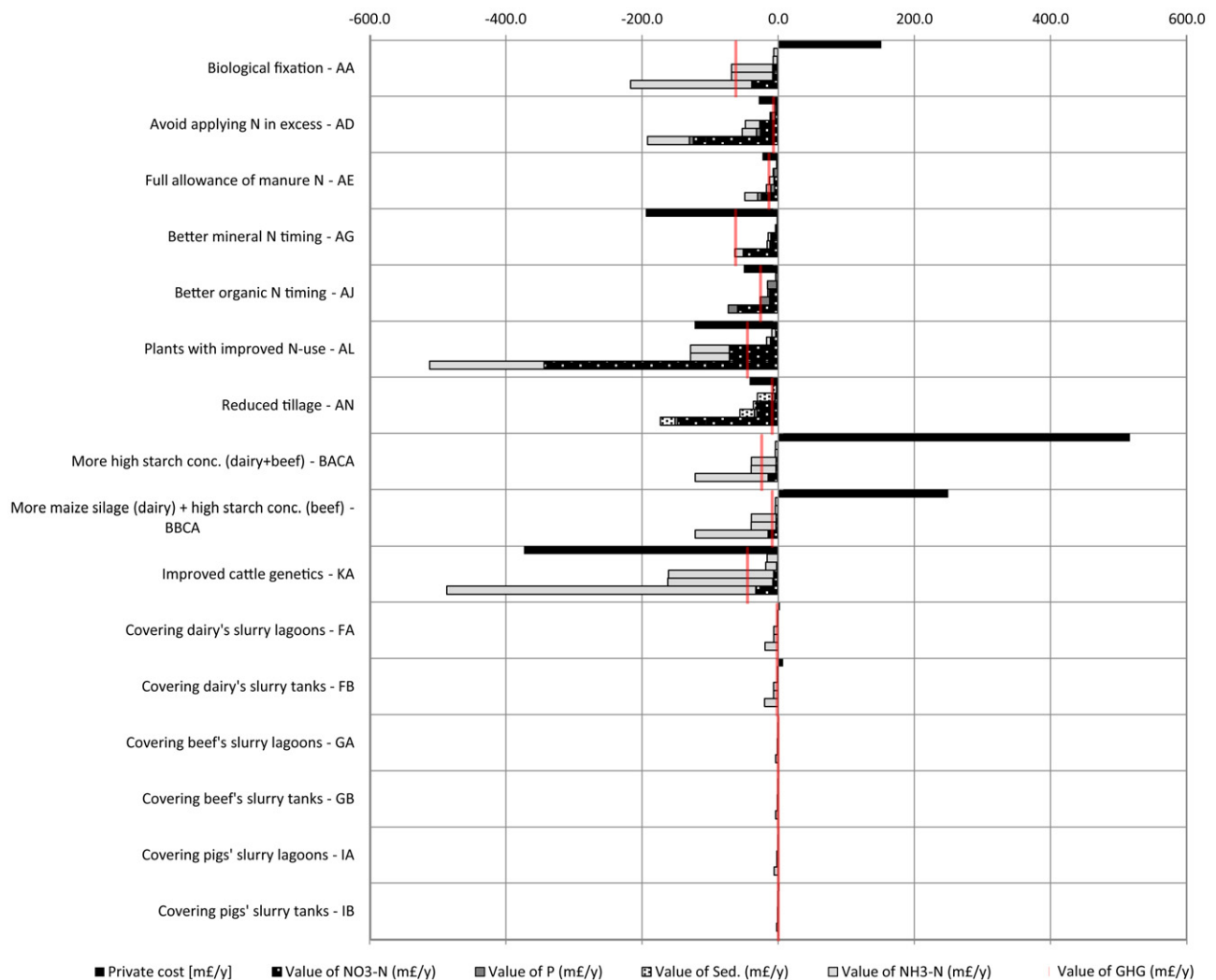
#### 4.1. Private and external cost comparison

Annual private cost of the measures fall in a range of £ –811 million (“Ionophores, beef”) to £ 1650 million (“Transgenic manipulation of ruminants, dairy”), negative values denoting a saving. Recall that this refers to the extent of cost/savings resulting from the application of a mitigation measure to its full extent, i.e. across the whole English and Welsh agriculture. The annual value of external effects varies from £ –16 million (“Improved cattle genetics”) to £ 0 (anaerobic digestion measures) calculated with damage cost set A, and from £ –512 million (“Plant varieties with improved N-use efficiency”) to £ 0 (anaerobic digestion measures) calculated with damage cost set E (Fig. 2). The external effects are net positive for measures where data are available, because none have any

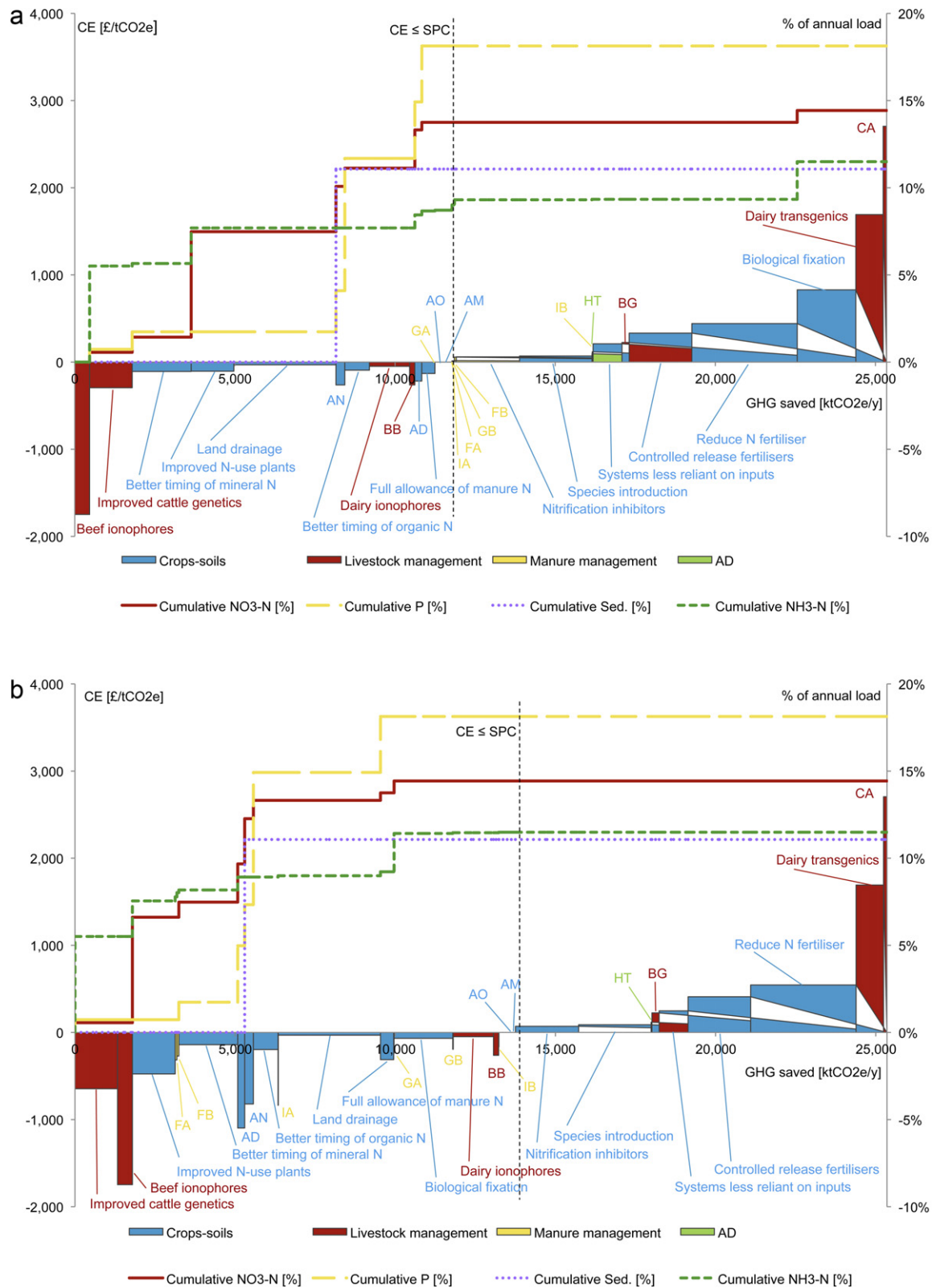
negative co-effects on the pollutants considered. The external effects have generally higher values for crop and soil measures and measures on cattle feeding and genetics, and are lower for manure management measures (covering slurry stores) and zero for anaerobic digestion measures. Changing between damage cost sets from A to E increases the amount of external benefits. The two biggest increases in the external benefits arise when changing from damage cost set B to C and damage cost set D to E, where  $\text{NH}_3$  and  $\text{NO}_3^-$  damage costs are increased considerably. Changing damage costs from set A to B or C to D (where the damage costs of phosphorous and sediment are increased) does not have a big impact on the value of co-effects.

#### 4.2. GHG-MACC

With no co-effects, the economically optimal GHG abatement (i.e. measures with CE below the shadow price of carbon), assuming full uptake of measures by the farmers, is



**Fig. 2 – Annual private and social costs and the value of GHG savings. Measures are shown where data are available on external effects (with the exception of anaerobic digestion measures, as the value of their co-effects are zero). The contribution of each co-effect to the social costs is indicated by separate colours. For each measure the six bars from top to bottom represent the private costs, social costs from damage cost set of A–E, respectively. The monetary value of the GHG savings (calculated by using the SPC) are shown by red vertical lines. Sed.: sediment. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)**



**Fig. 3 – GHG MACCs with co-effects valued with different damage cost sets: (a) MP-MACC-B, (b) MP-MACC-E. Sed.: sediment. See Appendix for the names of the measures with abatement potential less than 400 kt CO<sub>2</sub>e y<sup>-1</sup>.**

11.9 Mt CO<sub>2</sub>e y<sup>-1</sup> for England and Wales in 2022. The total abatement potential of measures with negative costs (i.e. measures' CE ≤ £0) is 11.8 Mt CO<sub>2</sub>e y<sup>-1</sup>. This is 36% of agricultural GHG emissions in England and Wales, which are expected to be 32.6 Mt CO<sub>2</sub>e in that year (Defra, 2011).

#### 4.3. MP-MACC-A and MP-MACC-B

Adding the effect of NO<sub>3</sub><sup>-</sup>, NH<sub>3</sub>, P and sediment valued with the damage cost sets A and B to the private costs of the measures has a small effect on the MACC (Fig. 3a). The cost-effectiveness improves slightly for all measures that have data on co-effects (except anaerobic digestion measures). The cumulative GHG abatement of measures with CE ≤ 0 increases only narrowly, by 14 kt CO<sub>2</sub>e y<sup>-1</sup>. Counter intuitively, the cumulative GHG abatement of measures with CE ≤ SPC is reduced by 110 kt CO<sub>2</sub>e y<sup>-1</sup>. This is due to on-farm pig anaerobic digestion measures being replaced by covering of slurry tanks and lagoons (pigs), as the latter became more cost-effective. Covering pig slurry tanks and lagoons provides smaller abatement potential than on-farm anaerobic digestion of pig manure.

MP-MACC-B shows only small differences from MP-MACC-A in spite of a three to five-fold increase in the damage cost of NO<sub>3</sub><sup>-</sup>, P and sediment, because the social benefits are still far less than the absolute value of the private costs/benefits.

For both damage cost sets A and B, the annual abatement potential under CE = 0 for the non-GHG pollutants are 38 kt NO<sub>3</sub><sup>-</sup>-N, 0.7 kt P, 198 kt sediment and 14 kt NH<sub>3</sub>-N (14%, 18%, 11% and 9% of annual load from agriculture, respectively). Total annual loads are estimated by Anthony et al. (2008) to be 276 kt nitrate, 4.0 kt phosphorous, 1790 kt sediment and 158 kt ammonia in 2020. The pollutant reduction results are the same when the CE is increased from 0 to the SPC.

#### 4.4. MP-MACC-C and MP-MACC-D

Applying higher damage costs, again leads to a slight change in the MACC, increasing the GHG abatement potential by 1% (MP-MACC-C or MP-MACC-D compared to MP-MACC-A or MP-MACC-B). The biggest difference from MP-MACC-A and MP-MACC-B is that the CEs of all but one of the slurry store covering MMs are now below the shadow price of carbon. Both the GHG and NH<sub>3</sub> savings are improved by this change (by

0.07 Mt CO<sub>2</sub>e and 0.4 kt NH<sub>3</sub>-N, respectively). On these MACCs, only two measures for which we have data on external effects remain economically inefficient (having CE > SPC): “Using biological fixation to provide N inputs” and “Centralised anaerobic digestion, poultry, 5MW”.

Again, the large increase in the value of P and sediment, in MP-MACC-D relative to MP-MACC-C has only a slight effect, improving the cost-effectiveness values of four measures that already had negative CEs.

#### 4.5. MP-MACC-E

With higher damage costs for NO<sub>3</sub><sup>-</sup> and NH<sub>3</sub> “Using biological fixation to provide N inputs” becomes economically efficient, with the cumulative annual GHG savings increasing by 1.8 Mt to 13.8 Mt CO<sub>2</sub>e, and the NH<sub>3</sub> savings by 3.4 kt to 18.2 kt NH<sub>3</sub>-N for both a CE equal to zero or the shadow price of carbon (Fig. 3b).

As shown in Fig. 4, introducing damage costs for measures where co-effect data are available improves the cost-effectiveness of all measures except for the anaerobic digestion measures, which have no effect on ammonia, nitrate, phosphorus or sediment pollution. The data available on external effects imply that no pollution swapping occurs.

## 5. Discussion

The results suggest that data on external effects can modify the MACC, but that the inclusion of external effects has surprisingly little impact on the cumulative pollution reductions in this analysis. Fully implementing cost-effective mitigation measures in England and Wales could save 37% of GHG emissions annually if no co-effects are included. MP-MACC-A to MP-MACC-D show the same GHG savings and 9–18% saving of the other four pollutants. Applying high damage costs (set E) drives up GHG, NO<sub>3</sub><sup>-</sup> and NH<sub>3</sub> savings by 6%, 1% and 2%, respectively (Table 3). As Table 3 shows, the overall effect on pollutant abatement is not greatly affected by the different damage costs considered in this study.

There are two main reasons for this. Comparing the monetary value of total pollution loads from agriculture (Table 4), it appears that with the conservative damage cost sets A and B the total negative impacts of non-GHG pollutants

**Table 3 – Pollution savings from GHG measures with CE < SPC (−34.3 £/tCO<sub>2</sub>e) in 2022 in England and Wales, assuming 100% uptake, expressed as % of business as usual agricultural load.**

MACC	GHG	NO <sub>3</sub> <sup>-</sup>	NH <sub>3</sub>	P	Sediment	Annual cost (£ million) <sup>a</sup>
GHG-MACC	36.7	n/a	n/a	n/a	n/a	−1841
MP-MACC-A	36.4	13.8	9.1	18.1	11.1	−1845
MP-MACC-B	36.4	13.8	9.1	18.1	11.1	−1845
MP-MACC-C	36.6	13.8	9.3	18.1	11.1	−1836
MP-MACC-D	36.6	13.8	9.3	18.1	11.1	−1836
MP-MACC-E	42.2	14.4	11.5	18.1	11.1	−1685
Business as usual annual load	32.6 Mt CO <sub>2</sub> e	276.1 kt NO <sub>3</sub> <sup>-</sup> -N	158.2 kt NH <sub>3</sub> -N	4.0 kt P	1790.8 kt sediment	

<sup>a</sup> Note that a negative cost implies saving.

**Table 4 – Value of agricultural pollution load of GHG, NO<sub>3</sub><sup>-</sup>, NH<sub>3</sub>, P and sediment in 2020 in England and Wales, £ million (Anthony et al., 2008).**

Damage cost set	GHG	NO <sub>3</sub> <sup>-</sup>	NH <sub>3</sub>	P	Sediment	Total
A	1118	60	285	39	44	1546
B		185	285	181	193	1962
C		1184	2800	385	44	5531
D		1184	2800	181	193	5475
E		5681	8235	181	193	15,408

GHG is valued at 34.3 £ tCO<sub>2</sub>e<sup>-1</sup> (Price et al., 2007).

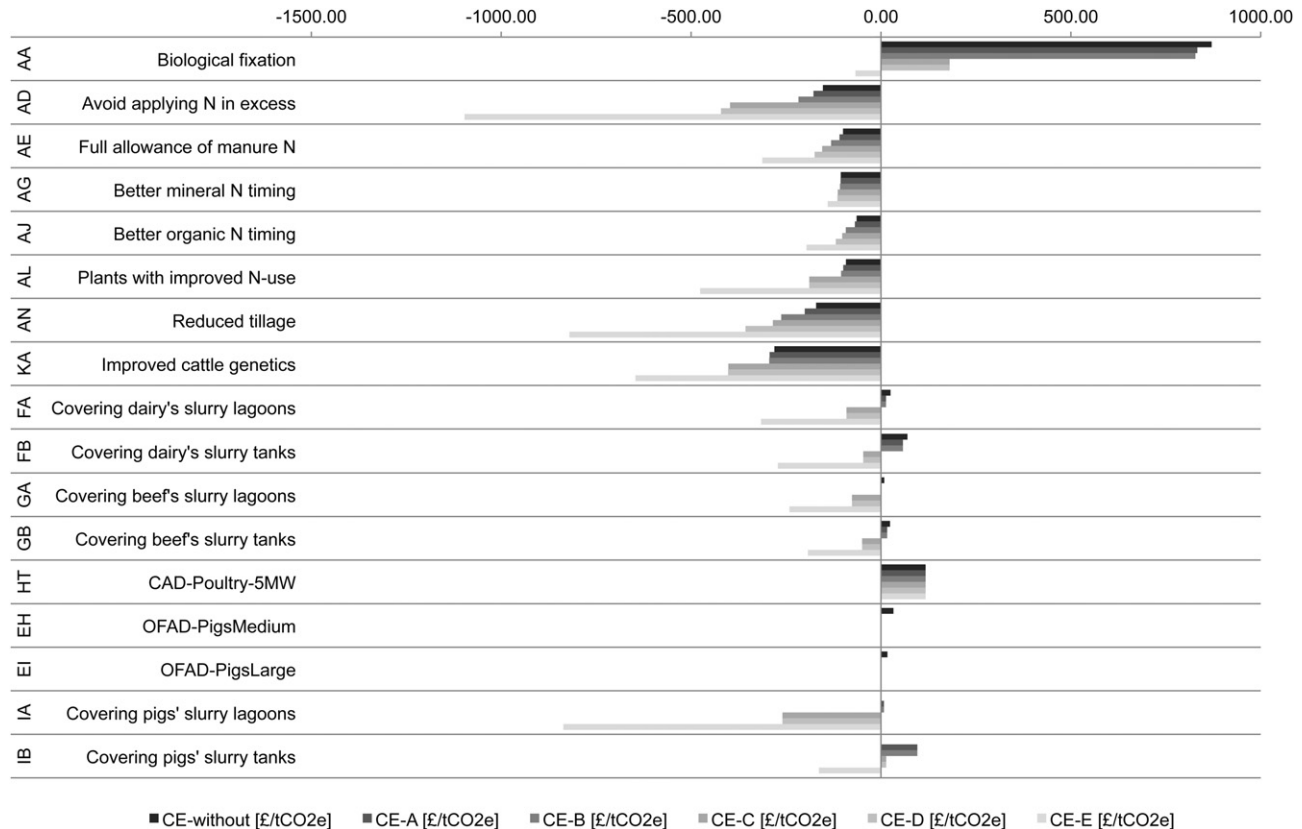
are substantially lower than of GHGs. Consequently, measures designed to reduce GHG emissions are likely to have much lower monetary impacts on non-GHG pollution than on GHG pollution. This is reflected in Fig. 2, where the monetary value of the mitigation measures' co-effects with damage cost sets A and B is generally 3–10 times lower than the monetary value of GHG savings.

Increasing the damage costs (set C) makes a difference and Table 4 shows that the value of both the nitrate and the ammonia total agricultural pollution is higher than the value of total agricultural GHG pollution. In parallel, for many mitigation measures the combined monetary values of co-effects are now 2–6 higher than the monetary value of the GHG savings (Fig. 2). This should bring those measures with private cost-effectiveness slightly above the shadow price of carbon, below this threshold. And it does (see measures “Covering

pigs' slurry tanks” and “Covering dairy cattle's slurry tanks” in Fig. 4.), but this is not the case for all the measures.

The second reason is the lack of data on external effects for many measures. Only three of the eleven measures for which CE > SPC on the GHG-MACC have data on co-effects, and only one of these has abatement potential considerable enough to change the shape of the MACC (“Using biological fixation to provide N inputs”, with abatement potential of 1.8 Mt CO<sub>2</sub>e y<sup>-1</sup>). However, this measure becomes efficient only when damage cost set E is applied.

The eight measures for which CE > SPC on the GHG-MACC and external data were not available can potentially change the economically optimal abatement level. Their cumulative annual GHG abatement potential is 11.6 MtCO<sub>2</sub>e – almost as much as the economically optimal abatement level on the GHG-MACC. It is possible that MMs “Nitrification inhibitors”



**Fig. 4 – Cost-effectiveness of measures on the MACCs which have data on co-effects. CE-without, CE-A, CE-B, CE-C, CE-D and CE-E represents cost-effectiveness values as calculated in GHG-MACC, MP-MACC-A, MP-MACC-B, MP-MACC-C, MP-MACC-D and MP-MACC-E, respectively. OFAD: on-farm anaerobic digestion, CAD: centralised anaerobic digestion.**

and “Species introduction” would become cost-effective by including their co-effects on nitrate leaching calculated by damage cost sets A and B, because their private CE is close to the threshold (59 and 69 £ tCO<sub>2</sub>e<sup>-1</sup>, respectively). This change would add further 4.2 Mt CO<sub>2</sub>e y<sup>-1</sup> to the GHG abatement potential under SPC. Damage cost sets C, D or E might increase the economically optimal abatement potential by an additional 6.4 Mt CO<sub>2</sub>e y<sup>-1</sup> due to the potentially reduced CE of “Reduce N fertiliser”, “Adopting systems less reliant on input” and “Controlled release fertilisers”. The extremely high CE of the measure “Transgenic manipulation of ruminants, dairy” might prevent it being a cost-effective measure even with the highest damage cost set applied.

On the other hand, there are some measures currently with no data on co-effects which might have negative effects on one or more of the other four pollutants. For example, “Use composts, straw-based manures in preference to slurry” might increase ammonia emissions from housing and storage (Chadwick et al., 2011; Jungbluth et al., 2001).

At least two further caveats should be noted relating to the costs included in this study. The first is that the current work attempts to add only four co-effects to the GHG MACC. It is clear that mitigation measures can have numerous other environmental impacts that should be taken into account. For example, reduced tillage can potentially increase pesticide, fungicide and insecticide use, affecting biodiversity and water quality. Cattle feeding measures requiring more grain could cause an expansion in the area of arable land, with the land use change having negative implications on biodiversity and beyond farm-gate GHG emissions. Reducing nitrogen fertilisation below the economic optimum would, again, provoke land use change. The issues of displaced production and full life-cycle costing of MMs are further critiques of existing MACCs, which we have not addressed in this paper.

The second cost issue concerns the avoided control costs related to the reduced non GHG emissions. Implementing a GHG measure might alter the total cost of other pollution control, and this change could be apportioned to GHG measures. In an ideal world, policy would be informed by pollutant-specific MACCs that encompass all relevant cost (including control costs) and benefits. These would have clearly delineated cost boundaries around measures and there would be agreement on where control costs lie. Since in practice the quantification of these costs would require complete knowledge about the other MACCs, and the apportionment might be highly debatable, the current study merely an attempt to apportion external cost impacts (related to GHG measures) to a GHG MACC, without adding the control cost implications the implemented GHG measures have on other pollutants. Because of the multiple effects if single measures, the total control costs of different pollutants cannot simply be added up without the risk of double counting, but should be calculated by adding up the costs of all the measures one by one.

In addition to MM uncertainties, the current study highlights further data needs in terms of the unit damage costs. While the difference between the lower and the higher carbon value estimates used in UK policy appraisal is three-fold, greater differences are revealed in existing estimates for the other damage costs. The complexity of agricultural externalities and the use of benefits transfer in valuation enhance uncertainties

in representing the damage functions for these pollutants. For a discussion of the difficulties associated with environmental valuation, see Smart et al. (2011). These uncertainties emphasise the importance of using cost-effectiveness and cost-benefit calculations as a complementary rather than exclusive policy tool, with a clear understanding of their advantages and limitations.

Finally, note that the choice of the carbon threshold will inevitably influence the estimated economically optimal abatement potential. We used the shadow price of carbon estimated by Price et al. (2007), while the UK Government current approach (based on reduction targets) values carbon in the non-traded sector in 2022 at 31–93 £ tCO<sub>2</sub>e<sup>-1</sup>, with a central value of 62 £ tCO<sub>2</sub>e<sup>-1</sup> (DECC, 2009). Using the DECC central value would move the “Nitrification inhibitors” MM into the efficient category on the GHG-MACC.

## 6. Conclusion

The omission of external effects has been highlighted as a drawback of GHG MACC analysis in policy making. The evidence presented here shows how the inclusion of external effects can alter the cost-effectiveness of environmental measures and how alternative damage cost estimates for nitrate, ammonia, phosphorus and sediment can change the results of abatement potential estimates derived in the 2022 MTP Optimistic GHG MACC. Higher damage cost values (sets C and D) make some measures more cost-effective, improving both the cumulative GHG abatement potential and the associated gains in the other four pollutants. Very high damage costs (set E) would justify the implementation of almost all the GHG measures which have positive co-effects. This finding is in line with the estimated costs of different pollutants originating from agriculture: using lower damage costs (sets A and B) the total cost is dominated by GHG emissions, increasing the non-GHG pollutants' damage costs to set E shrinks to GHG's contribution to costs to the tenth of its original share.

This study highlights the gaps in data availability for other externalities relevant to GHG mitigation measures. Ongoing and future experimental and modelling research should focus on expanding the scope of research beyond GHG effects, especially in relation to the mitigation measures with high abatement potential, like nitrification inhibitors, controlled release fertilisers, species introduction and systems less reliant on input. Advances in monetary valuation of pollutants are also desired. Notwithstanding the data gaps, the MP MACC is a useful analytical device for cumulating knowledge about the GHG efficiency, co-effects and private costs of GHG mitigation measures, and offers easy visual representation of the integrated information.

The multiple pollutant MACC can offer specific policy messages for agencies trying to interpret MACC information. The first is to focus any further analysis on GHG MMs that are slightly above the threshold on the GHG MACCs, as they most probably have co-effects which could make their implementation worthwhile. The second message is to explore thoroughly any possible negative external effects of those GHG measures that are cost-effective on the GHG MACCs and



become cost-effective on the MP MACCs. In these cases it may be useful to consider effects beyond those analysed in this paper, like biodiversity, soil quality, human health, animal health and welfare and social effects (e.g. food security, resilience of rural communities).

## Acknowledgements

This analysis has been undertaken with Scottish Government Rural and Environmental Science and Analytical Services division (RESAS) funding to SAC and to ClimatexChange

(<http://www.climatexchange.org.uk/>). Part of the research leading to these results has been conducted as part of the AnimalChange project which received funding from the European Community's Seventh Framework Programme (FP7/2007–2013) under the grant agreement no. 266018. The authors thank the anonymous referees for very useful comments on a previous draft.

## Appendix

See Table A.1.

**Table A.1 – Matching measures between the two studies (Anthony et al., 2008; MacLeod et al., 2010b). ID refers to the measures' codes in these studies, respectively.**

(MacLeod et al., 2010b)		(Anthony et al., 2008)	
ID	Measure name	ID	Measure name
AA	Using biological fixation to provide N inputs (clover)	31	Use clover in place of grass
AD	Avoiding N excess	22	Use a fertiliser recommendation system
AE	Full allowance of manure N supply	23	Integrate fertiliser and manure nutrient supply
AG	Improved timing of mineral fertiliser N application	26	Avoid spreading fertiliser to fields at high-risk times
AJ	Improved timing of slurry and poultry manure application	71	Do not spread slurry at high-risk times
		74	Do not spread solid manure to fields at high-risk times Results added up
AL	Plant varieties with improved N-use efficiency	20	Use plants with improved nitrogen use efficiency
AN	Reduced tillage/No-till	7	Adopt reduced cultivation systems
BA	Increased high starch concentrate in diet, dairy	33	Reduce dietary nitrogen and phosphorus intakes
BB	Increased maize silage in diet, dairy	Notes: 1. The P-effect of Measure 33 was ignored 2. Most pig and poultry farms already use high quality protein feeds, therefore Measure 33 mainly refer to changes in cattle feeding 3. In cattle the main methods of reducing the N content of the diet are increasing the starch and reducing the grass/forage content of the feed by increasing the proportion of starchy concentrates or maize silage	
CA	Increased high starch concentrate in diet, beef		
Merged into two new measures to cover the all cattles (note that BA and BB are mutually exclusive):			
BACA	Increased high starch concentrate in diet, dairy + beef		
BBCA	Increased maize silage for dairy and increased high starch concentrate for beef		
BF	Improved genetic potential for dairy cows–productivity	19	Make use of improved genetic resources in livestock
BI	Improved genetic potential for dairy cows–fertility		
CG	Improved genetic potential for beef cattle		
Merged into a new measure:			
KA	Improved cattle genetics		
FA	Covering lagoons – dairy	56	Install covers to slurry stores
FB	Covering slurry tanks – dairy	Results disaggregated into dairy/beef/pig tank/lagoon categories	
GA	Covering lagoons – beef		
GB	Covering slurry tanks – beef		
IA	Covering lagoons – pigs		
IB	Covering slurry tanks – pigs		
EB-EI	On-farm anaerobic digestion (OFAD) measures (dairy/beef/pig, medium/large farms)	58	Anaerobic digestion
HA-HT	Centralised anaerobic digestion (CAD) measures (dairy/beef/pigs/poultry, 1MW/2MW/3MW/4MW/5MW)		
AB	Reduce N fertiliser	–	No matching measure
AC	Land drainage	–	No matching measure

**Table A.1 (Continued)**

(MacLeod et al., 2010b)		(Anthony et al., 2008)	
ID	Measure name	ID	Measure name
AF	Species introduction (including legumes)	–	No matching measure
AH	Controlled release fertilisers	–	No matching measure
AI	Nitrification inhibitors	–	No matching measure
AK	Adopting systems less reliant on inputs	–	No matching measure
AM	Separate slurry applications from fertiliser applications by several days	–	No matching measure
AO	Use composts, straw-based manures in preference to slurry	–	No matching measure
BE	Ionophores, dairy	–	No matching measure
BG	Bovine somatotropin, dairy	–	No matching measure
BC	Propionate precursors, dairy	–	No matching measure
BD	Probiotics, dairy	–	No matching measure
BH	Transgenic manipulation of ruminants, dairy	–	No matching measure
CC	Propionate precursors, beef	–	No matching measure
CD	Probiotics, beef	–	No matching measure
CE	Ionophores, beef	–	No matching measure
FC	Switch from anaerobic to aerobic storage – dairy slurry tanks	–	No matching measure
FD	Switch from anaerobic to aerobic storage – dairy lagoons	–	No matching measure
GC	Switch from anaerobic to aerobic storage – beef slurry tanks	–	No matching measure
GD	Switch from anaerobic to aerobic storage – beef lagoons	–	No matching measure
IC	Switch from anaerobic to aerobic storage – pigs slurry tanks	–	No matching measure
ID	Switch from anaerobic to aerobic storage – pigs lagoons	–	No matching measure

## REFERENCES

- Anthony, S., Duethman, D., Gooday, R., Harris, D., Newell-Price, P., Chadwick, D., Misselbrook, T., 2008. Quantitative Assessment of Scenarios for Managing Trade-Off Between Economics, Environment and Media. Report No. Defra WQ0106 (Module 6). Defra, ADAS, North Wyke Research.
- Baker, B., Metcalfe, P., Butler, S., Gueron, Y., Sheldon, R., East, J., 2007. The Benefits of Water Framework Directive Programmes of Measures in England and Wales. Report No. Defra CRP 4b/c. Defra, NERA, Accent.
- Brink, C., van Grinsven, H., Jacobsen, B.H., Rabl, A., Gren, I.M., Holland, M., Klimont, Z., Hicks, W.K., Brouwer, R., Dickens, R., Willems, J., Termansen, M., Velthof, G.L., Alkemade, R., van Oorschot, M., Webb, J., 2011. Costs and benefits of nitrogen in the environment. In: Sutton, M.A., Howard, C.M., Erisman, J.W., Billen, G., Bleeker, A., Grennfelt, P., Grinsven, H., Grizzetti, B. (Eds.), *The European Nitrogen Assessment*. 1st ed. Cambridge University Press, Cambridge, pp. 513–540.
- Brink, C., Kroeze, C., Klimont, Z., 2001. Ammonia abatement and its impact on emissions of nitrous oxide and methane – part 2: application for Europe. *Atmospheric Environment* 35, 6313–6325.
- Brink, C., van Ierland, E., Hordijk, L., Kroeze, C., 2005. Cost-effective emission abatement in agriculture in the presence of interrelations: cases for the Netherlands and Europe. *Ecological Economics* 53, 59–74.
- Brouwer, R., 2000. Environmental value transfer: state of the art and future prospects. *Ecological Economics* 32, 137–152.
- Chadwick, D., Misselbrook, T., Gilhespy, S., Williams, J., Bhogal, A., Sagoo, L., Nicholson, F., Webb, J., Anthony, S., Chambers, B., 2005. Ammonia emissions and crop N use efficiency to grassland and tillage land; factors affecting ammonia emissions from urea based fertilisers; Ammonia emissions model. Report No. Defra NT2605 WP1b. Defra, IGER, ADAS.
- Chadwick, D., Sommer, S., Thorman, R., Fangueiro, D., Cardenas, L., Amon, B., Misselbrook, T., 2011. Manure management: implications for greenhouse gas emissions. *Animal Feed Science and Technology* 166–167, 514–531.
- Chambers, B.J., Lord, E.I., Nicholson, F.A., Smith, K.A., 1999. Predicting nitrogen availability and losses following application of organic manures to arable land: MANNER. *Soil Use and Management* 15, 137–143.
- Correll, D.L., 1998. The role of phosphorus in the eutrophication of receiving waters: a review. *Journal of Environment Quality* 27, 261–266.
- Davison, P.S., Withers, P.J., Lord, E.I., Betson, M.J., Stroemqvist, J., 2008. PSYCHIC – a process-based model of phosphorus and sediment mobilisation and delivery within agricultural catchments. Part 1: model description and parameterisation. *Journal of Hydrology* 350, 290–302.
- DECC, 2009. Carbon Valuation in UK Policy Appraisal: A Revised Approach. *Climate Change Economics*. Department of Energy and Climate Change (DECC), London.
- Defra, 2008. Damage cost estimates. <http://archive.defra.gov.uk/environment/quality/air/airquality/panels/igcb/guidance/damagecosts.htm>
- Defra, 2011. Greenhouse Gas Emission Projections for UK Agriculture to 2030. Report No. PB 13622. Defra, London.
- Enkvist, P.A., Naucler, T., Rosander, J., 2007. A cost curve for greenhouse gas reduction. McKinsey.
- Georgiou, S., Bateman, I., Cole, M., Hadley, David, 2000. Contingent Ranking and Valuation of River Water Quality Improvements: Testing for Scope Sensitivity, Ordering and Distance Decay Effects. Report No. CSERGE Working Paper GEC 2000-18.
- Gren, I.M., Jonzon, Y., Lindqvist, M., 2008. Costs of nutrient reductions to the Baltic Sea. Technical Report. Report No. Working Paper Series 2008:1. Swedish University of Agricultural Sciences (SLU), Department of Economics, Uppsala.
- Haygarth, P., ApSimon, H., Betson, M., Harris, D., Hodgkinson, R., Withers, P., 2009. Mitigating diffuse phosphorus transfer from agriculture according to cost and efficiency. *Journal of Environment Quality* 38, 2012–2022.
- Holland, M., Pye, S., Watkiss, P., Droste-Franke, B., Bickel, P., 2005. Damages per tonne emission of PM<sub>2.5</sub>, NH<sub>3</sub>, SO<sub>2</sub>, NO<sub>x</sub>

- and VOCs from each EU25 Member State (excluding Cyprus) and surrounding seas. AEA Technology Environment; European Commission DG Environment.
- IGER, 2001. Cost curve assessment of mitigation options in greenhouse gas emissions from agriculture. Report No. Defra CC0229. Defra, London.
- Jungbluth, T., Hartung, E., Brose, G., 2001. Greenhouse gas emissions from animal houses and manure stores. *Nutrient Cycling in Agroecosystems* 60, 133–145.
- Kesicki, F., Strachan, N., 2011. Marginal abatement cost (MAC) curves: confronting theory and practice. *Environmental Science and Policy* 14, 1195–1204.
- Lord, E.I., 1992. Modelling of nitrate leaching: Nitrate Sensitive Areas. *Aspects of Applied Biology* 30, 19–28.
- Lord, E.I., Anthony, S.G., 2000. MAGPIE: a modelling framework for evaluating nitrate losses at national and catchment scales. *Soil Use and Management* 16, 167–174.
- MacLeod, M., Moran, D., Eory, V., Rees, R.M., Barnes, A., Topp, C.F.E., Ball, B., Hoad, S., Wall, E., McVittie, A., Pajot, G., Matthews, R., Smith, P., Moxey, A., 2010a. Developing greenhouse gas marginal abatement cost curves for agricultural emissions from crops and soils in the UK. *Agricultural Systems* 103, 198–209.
- MacLeod, M., Moran, D., McVittie, A., Rees, R., Jones, G., Harris, D., Antony, S., Wall, E., Eory, V., Barnes, A., Topp, Cairistiona F.E., Ball, Bruce, Hoad, Steve, Eory, L., 2010. Review and Update of UK Marginal Abatement Cost Curves for Agriculture. Report No. Report to CCC.
- Moran, D., MacLeod, M., Wall, E., Eory, V., McVittie, A., Barnes, A., Rees, R., Topp, C.F.E., Moxey, A., 2011a. Marginal abatement cost curves for UK agricultural greenhouse gas emissions. *Journal of Agricultural Economics* 62, 93–118.
- Moran, D., MacLeod, M., Wall, E., Eory, V., McVittie, A., Barnes, A., Rees, R.M., Topp, C.F.E., Pajot, G., Matthews, R., Smith, P., Moxey, A., 2011b. Developing carbon budgets for UK agriculture land-use, land-use change and forestry out to 2022. *Climatic Change* 105, 529–553.
- Moran, D., MacLeod, M., Wall, E., Eory, V., Pajot, G., Matthews, R., McVittie, A., Barnes, A., Rees, R., Moxey, A., Williams, A., Smith, P., 2008. UK Marginal Abatement Cost Curves for the Agriculture and Land Use, Land-Use Change and Forestry Sectors out to 2022. With Qualitative Analysis of Options to 2050. Report No. RMP4950, Committee on Climate Change. SAC.
- O'Neill, D., 2007. The Total External Environmental Costs and Benefits of Agriculture in the UK. EPA.
- Oenema, O., Salomez, J., Branquinho, C., Budnakova, M., Cermak, P., Geupel, M., Johnes, P., Tompkins, C., Spranger, T., Erisman, J.W., Palliere, C., Maene, L., Alonso, R., Maas, R., Magid, J., Sutton, M.A., van Grinsven, H., 2011. Developing integrated approaches to nitrogen management. In: Sutton, M.A., Howard, C.M., Erisman, J.W., Billen, G., Bleeker, A., Grennfelt, P., Grinsven, H., Grizzetti, B. (Eds.), *The European Nitrogen Assessment*. 1st ed. Cambridge University Press, Cambridge, pp. 541–550.
- Pearce, D.W., Turner, R.K., 1990. Economics of natural resources and the environment. Harvester Wheatsheaf, Great Britain.
- Pimentel, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Shpritz, L., Fitton, L., Saffouri, R., Blair, R., 1995. Environmental and economic costs of soil erosion and conservation benefits. *Science* 267, 1117–1123.
- Pretty, J.N., Mason, C.F., Nedwell, D.B., Hine, R.E., 2002. A Preliminary Assessment of the Environmental Costs of the Eutrophication of Fresh Waters in England and Wales. University of Essex, Colchester UK.
- Price, R., Thornton, S., Nelson, S., 2007. The Social Cost of Carbon and the Shadow Price of Carbon: What They Are, and How to Use them in Economic Appraisal in the UK. Department for Environment, Food and Rural Affairs, Crown copyright, London.
- Reis, S., Nitter, S., Friedrich, R., 2005. Innovative approaches in integrated assessment modelling of European air pollution control strategies – implications of dealing with multi-pollutant multi-effect problems. *Environmental Modelling and Software* 20, 1524–1531.
- Schneider, U.A., Mccarl, B.A., Schmid, E., 2007. Agricultural sector analysis on greenhouse gas mitigation in US agriculture and forestry. *Agricultural Systems* 94, 128–140.
- Scholefield, D., Haygarth, P., 2004. Cost curve of nitrate mitigation options. Report No. Defra NT2511. Defra, London.
- Shepherd, M.A., Anthony, S., Temple, M., Burgess, D., Patton, M., Renwick, A., Barnes, A., Chadwick, D., 2007. Baseline Projections for Agriculture and Implications for Emissions to Air and Water. Report No. Defra SFF0601. Defra, ADAS, SAC, IGER, London.
- Smart, J.C., Hicks, K., Morrissey, T., Heinemeyer, A., Sutton, M.A., Ashmore, M., 2011. Applying the ecosystem service concept to air quality management in the UK: a case study for ammonia. *Environmetrics* 22, 649–661.
- Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., 2007. Agriculture. In: Metz, B., Davidson, O.R., Bosch, P.R., Dave, R., Meyer, L.A. (Eds.), *Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
- Soderqvist, T., Hasselstrom, L., 2008. The economic value of ecosystem services provided by the Baltic Sea and Skagerrak. Report No. Swedish Environmental Protection Agency 5874. Swedish Environmental Protection Agency.
- Spencer, I., Bann, C., Moran, D., McVittie, A., Lawrence, K., Caldwell, V., 2008. Environmental Accounts for Agriculture. Report No. Defra SFS0601. Defra, Welsh Assembly Government, Scottish Government, Department of Agriculture and Rural Development (N. Ireland), SAC, Cranfield University.
- Thomas, J., Thistlethwait, G., MacCarthy, J., Pearson, B., Murrels, T., Pang, Y., Passant, N., Webb, N., Connolly, C., Cardenas, L., Malcolm, H., Thomson, A., 2011. Greenhouse Gas Inventories for England, Scotland, Wales and Northern Ireland: 1990–2009. Report No. AEAT/ENV/R/3222 Issue 1, ED56403200/Issue 1. AEA.
- van Grinsven, H.J., Rabl, A., de Kok, T.M., 2010. Estimation of incidence and social cost of colon cancer due to nitrate in drinking water in the EU: a tentative cost-benefit assessment. *Environmental Health* 9, 58.
- Wagner, F., Amann, M., Borken-Kleefeld, J., Cofala, J., Hoglund-Isaksson, L., Purohit, P., Rafaj, P., Schopp, W., Winiwarter, W., 2012. Sectoral marginal abatement cost curves: implications for mitigation pledges and air pollution co-benefits for Annex I countries. *Sustainability Sciences* 7, 169–184.
- Webb, J., Misselbrook, T.H., 2004. A mass-flow model of ammonia emissions from UK livestock production. *Atmospheric Environment* 38, 2163–2176.
- Webb, J., Ryan, M., Anthony, S.G., Brewer, A., Laws, J., Aller, M.F., Misselbrook, T.H., 2006. Cost-effective means of reducing ammonia emissions from UK agriculture using the NARSES model. *Atmospheric Environment* 40, 7222–7233.

Vera Eory is a Climate Change Researcher at SRUC. Her research focuses on the economic and environmental implications of greenhouse gas reduction practices in agriculture. Besides, she



is interested in the uncertainties around mitigation estimates, farmers' perception on GHG mitigation, and joint modeling of the biophysical and the economic aspects of pollution reduction.

**Cairistiona F.E. Topp** research interests include assessing the global warming potential of different management systems, mitigation and adaptation strategies for farming systems, and tools to compare the sustainability of different farming systems. Her areas of expertise also include understanding the impact of changes in farm management in cropping and grassland systems on the

environment, and in how an understanding of these issues can inform economic and social models of the farming system.

**Dominic Moran** is Professor of Environmental Economics at SRUC. He is also currently a DFID Senior Research Fellow with the Climate and Environment Research Group. He specialises in applied cost-benefit analysis of environmental and agri-environmental policy. This includes the economics of biodiversity conservation and climate change (mitigation and adaptation in agriculture and related land use).

# Linking an Economic and a Life-cycle Analysis Biophysical Model to Support Agricultural Greenhouse Gas Mitigation Policy

## Kombination eines ökonomischen Modells mit einem bio-physikalischen Lebenszyklus-Modell zur Unterstützung von Politikmaßnahmen zur Verringerung von Treibhausgasen

Vera Eory, Michael MacLeod, Shailesh Shrestha and David Roberts  
Scotland's Rural College (SRUC), Edinburgh, UK

### Abstract

*Greenhouse gas (GHG) mitigation is one of the main challenges facing agriculture, exacerbated by the increasing demand for food, in particular for livestock products. Production expansion needs to be accompanied by reductions in the GHG emission intensity of agricultural products, if significant increases in emissions are to be avoided. Suggested farm management changes often have systemic effects on farm, therefore their investigation requires a whole farm approach. At the same time, changes in GHG emissions arising off-farm in food supply chains (pre- or post-farm) can also occur as a consequence of these management changes. A modelling framework that quantifies the whole-farm, life-cycle effects of GHG mitigation measures on emissions and farm finances has been developed. It is demonstrated via a case study of sexed semen on Scottish dairy farms. The results show that using sexed semen on dairy farms might be a cost-effective way to reduce emissions from cattle production by increasing the amount of lower emission intensity 'dairy beef' produced. It is concluded that a modelling framework combining a GHG life cycle analysis model and an economic model is a useful tool to help designing targeted agri-environmental policies at regional and national levels. It has the flexibility to model a wide variety of farm types, locations and management changes, and the LCA-approach adopted helps to ensure that GHG emission leakage does not occur in the supply chain.*

### Key words

*greenhouse gas mitigation; dairy farms; marginal abatement cost curves; life cycle analysis; whole farm modelling*

### Zusammenfassung

*Die Verringerung der Emissionen von Treibhausgasen (THG) ist eine der wichtigsten Herausforderungen für die Landwirtschaft, vor allem wegen der steigenden Nachfrage nach Lebensmitteln, insbesondere für tierische Erzeugnisse. Eine Ausweitung der Produktion muss von einer Verringerung der THG-Emissionsintensität landwirtschaftlicher Erzeugnisse begleitet werden, um die Zunahme von Emissionen zu vermeiden. Änderungen im Management wirken oft auf den ganzen landwirtschaftlichen Betrieb. Die Untersuchung hat diesem Umstand Rechnung zu tragen. Änderungen der THG-Emissionen in vor- und nachgelagerten Bereichen können auf Veränderungen im Management landwirtschaftlicher Betriebe zurückzuführen sein. Im Beitrag wird ein Modellierungszugang vorgestellt, der den gesamten Betrieb, den Lebenszyklus der Produkte und Auswirkungen der THG-Minderungsmaßnahmen auf Emissionen und wirtschaftliche Erfolgsgrößen quantifiziert. In der Fallstudie werden Auswirkungen des Einsatzes von gesextem Sperma in schottischen Milchviehbetrieben untersucht. Die Analyse zeigt, dass gesextes Sperma ein kostengünstiger Weg ist, um die Emissionen in der Rinderproduktion zu senken, und zwar durch geringere Emissionsintensität der Kuppelprodukte Milch - Rindfleisch. Die Ergebnisse zeigen den Vorzug eines Modellierungsansatzes in dem eine THG-Lebenszyklus-Analyse und ein Betriebsmodell kombiniert werden. Dies kann dazu dienen, Maßnahmen der Agrarumweltpolitik auf regionaler und nationaler Ebene gezielt einzusetzen. Der Zugang verfügt über die Flexibilität, eine Vielzahl von Betriebstypen, Standorte und Management-Veränderungen zu modellieren. Die Lebenszyklus-Analyse hilft, allfällige THG-Leckage-Effekte in der Versorgungskette aufzudecken.*

## Schlüsselwörter

*Treibhausgasemission; Milchviehhaltung; Grenzvermeidungskosten; Lebenszyklus-Analyse; landwirtschaftliche Betriebsmodellierung*

## List of Abbreviations

CH <sub>4</sub>	methane
CO <sub>2</sub>	carbon dioxide
EI	emission intensity
FAO	UN Food and Agriculture Organisation
FAS	Farm Account Survey of Scotland
GHG	greenhouse gases
GLEAM	Global Livestock Environmental Assessment Model
IPCC	Intergovernmental Panel on Climate Change
LCA	life cycle analysis
LP	linear programming
N	nitrogen
N <sub>2</sub> O	nitrous oxide

## 1 Introduction

Reducing greenhouse gas emissions arising from agricultural activities remains a challenge as the world is starting to experience the consequences of a changing climate (IPCC, 2013) and at the same time food production is facing major challenges both in demand for land-based products and also in terms of production constraints (FORESIGHT, 2011). Satisfying growing demand for livestock products will lead to significant increases in the greenhouse gas emissions from the sector unless the emission intensity (i.e. the GHG emissions arising from the production of a unit of output, e.g. kg CO<sub>2</sub>e (litres of milk)<sup>-1</sup>) can be reduced.

Globally, cattle milk is the largest source of livestock protein and global milk demand is forecast to increase by 80% by 2050, relative to 2005/7 demand (ALEXANDRATOS and BRUINSMA, 2012). The greenhouse gas emissions arising from global milk production were quantified by GERBER et al. (2010) and increasing attention is being paid to finding ways of reducing the emission intensity of milk production.

Numerous management changes and technologies have been proposed to reduce on-farm emissions from livestock (see for example BELLARBY et al., 2013; COTTLE et al., 2011; HRISTOV et al., 2013). A few measures only affect one emission source on the

farm; for example reducing excess nitrogen fertiliser decreases nitrous oxide emission without any further implications on the other activities on farm. However, many measures can have system-wide effects, e.g. changing the ration can lead to changes in enteric methane emissions, changes in volatile solid and N excretion rates (with consequent impacts on manure CH<sub>4</sub> and N<sub>2</sub>O emissions), and also changes in the amount of meat or milk produced. The use of whole farm modelling approaches provides a powerful tool for analysing the system-wide effects of GHG mitigation measures on emissions and farm financial performance.

In addition to the systemic effects within the farm outlined above, interactions can also occur along the supply chain. For example, changing the way in which inputs such as synthetic fertilisers and feed materials are produced can change the emission intensities of the final commodities produced. These effects can be captured by using a life cycle analysis approach in the evaluation of mitigation measures.

Various whole farm models and modelling frameworks have been developed, mostly for one or two particular farming systems (see reviews of the ruminant systems by CROSSON et al., 2011, and SCHILS et al., 2007), while some are capable of simulating different farming systems (LOUHICHI et al., 2010; NEUFELDT and SCHAFFER, 2008). However, LCA GHG calculations are rarely provided by these tools, therefore in this paper we outline an approach which is capable of simulating management changes on various farm systems to provide ex-ante evaluation of LCA GHG emissions and economic effects.

The farm level modelling framework presented here consists of the Global Livestock Environmental Assessment Model, a life-cycle GHG emission model (MACLEOD et al., 2013) and ScotFarm, an optimising farm level model based on a linear programming farm economic model described by SHRESTHA (2004). Within this framework, the emissions, production and farm income can be calculated with and without mitigation measures, thus enabling the cost-effectiveness of measures and the interactions between the measures to be quantified for specific-farm systems and locations.

This paper provides an explanation of the approach and a case study of sexed semen on Scottish dairy farms. Finally, the strengths and weaknesses of the approach and options for future development are discussed.

## 2 Methodology

### 2.1 GLEAM

GLEAM is an LCA model developed by the UN Food and Agriculture Organisation (MACLEOD et al., 2013). It simulates processes within livestock production systems in order to assess their environmental performance. The current version of the model (V1.0) focuses primarily on the quantification of GHG emissions and includes: (a) pre-farm emissions arising from the manufacture of inputs; (b) on-farm emissions during crop and animal production; and (c) post-farm emissions arising from the processing and transportation of products to the retail point. Emissions and food losses that arise after delivery to the retail point are not included. While gases of minor importance have been omitted, the three major GHG in agriculture are included, namely: (1) methane (mainly from enteric fermentation, manure storage and rice cultivation), (2) nitrous oxide (from soils and manure storage) and (3) carbon dioxide from (a) the combustion of fossil fuels on-farm (e.g. in tractors and generators) and off-farm (in the manufacture of inputs, including mineral fertilisers, and in post-farm processing and transport) and (b) land use change. Carbon dioxide from the short biological cycles such as respiration and aerobic decomposition are not included. GLEAM calculates:

- total production of the main livestock commodities, i.e. meat, milk and eggs
- the total greenhouse gas emissions arising from that production
- the emission intensity of each commodity.

A brief overview of the model elements is given below, and values for selected parameters given in Table 1.

The **herd module** starts with the total number of animals of a given species and system. It determines the herd structure (i.e. the number of animals in each cohort, and the rate at which animals move between cohorts) and the characteristics of the average animal in each cohort (e.g. weight and growth rate).

The **manure module** calculates the rate at which total excreted N is applied to crops, accounting for losses during storage.

The **feed module** calculates key feed parameters, i.e. the nutritional content and emissions per kg of the feed ration.

The **system module** calculates each animal cohort's energy requirement, and the total amount of meat, milk and eggs produced each year. It also calculates the total annual emissions arising from manure management, enteric fermentation and feed production.

The **allocation module** combines the emissions from the system module with the emissions calculated outside GLEAM, i.e. emissions arising from (a) direct on-farm energy use; (b) the construction of farm buildings and manufacture of equipment; and (c) post-farm transport and processing. The total emissions are then allocated to the co-products (e.g. meat and milk) and the EI of the commodities are calculated.

### 2.2 ScotFarm

ScotFarm, a profit optimising financial model developed at SRUC, is based on a farm level dynamic line-

**Table 1. Value of selected parameters for lactating cows**

Category	Parameter	Value	Notes
<b>Ration</b>	Ration digestibility (%)	78	Calculated, based on a ration of 62% fresh grass, 38% compound feed
<b>Ration</b>	Ration emissions intensity (kg CO <sub>2</sub> e (kg DM) <sup>-1</sup> )	1.4	Calculated using IPCC (2006) Tier 1
<b>Intake</b>	NE requirement (MJ cow <sup>-1</sup> day <sup>-1</sup> )	121.8	Calculated using IPCC (2006) Tier 2
<b>Intake</b>	Feed intake (kg DM cow <sup>-1</sup> day <sup>-1</sup> )	15.4	Calculated using IPCC (2006) Tier 2
<b>Output</b>	Volatile solid excretion (VSx) (kg cow <sup>-1</sup> day <sup>-1</sup> )	3.64	Calculated using IPCC (2006) Tier 2
<b>Output</b>	N excretion (kg N cow <sup>-1</sup> day <sup>-1</sup> )	0.39	Calculated using IPCC (2006) Tier 2
<b>Output</b>	Enteric methane (kg CH <sub>4</sub> cow <sup>-1</sup> year <sup>-1</sup> )	109	Calculated using IPCC (2006) Tier 2
<b>Manure</b>	Methane conversion factor (% of VSx)	6.3	Calculated using IPCC (2006) Tier 2, based on 68% PRP, 32% slurry (no cover)
<b>Manure</b>	Manure methane (kg CH <sub>4</sub> cow <sup>-1</sup> year <sup>-1</sup> )	13.4	Calculated using IPCC (2006) Tier 2
<b>Other</b>	Average annual temperature (°C)	10	Assumption
<b>Other</b>	Methane conversion factor (Ym) (%)	6.5	IPCC (2006, Table 10.12)
<b>Other</b>	B <sub>0</sub> (m <sup>3</sup> CH <sub>4</sub> (kg VS) <sup>-1</sup> )	0.24	IPCC (2006, Table 10.A4)

Source: authors

ar programming model which is described in detail in SHRESTHA (2004). Modified versions of farm level linear programming models have been used in a number of farm level analyses of Irish agriculture (SHRESTHA and HENNESSY, 2006; SHRESTHA et al., 2007, 2013; HENNESSY et al., 2008). ScotFarm assumes that all farmers are profit oriented and maximise farm net income within a set of limiting farm resources. It consists of four production systems; dairy, beef, sheep and arable. These systems are constrained by the land, labour, feed and stock replacement available to a farm. The total land available to a farm is fixed. Farms are allowed to buy in feeds, animal replacements and hire labour if required. The farm net income is comprised of the accumulated revenues collected from the final product of the farm activities (crops, animals and milk) plus farm payments minus costs incurred for inputs under those activities. The input costs are replacement costs for livestock, variable costs including labour, feed and veterinary costs and overhead costs on farms.

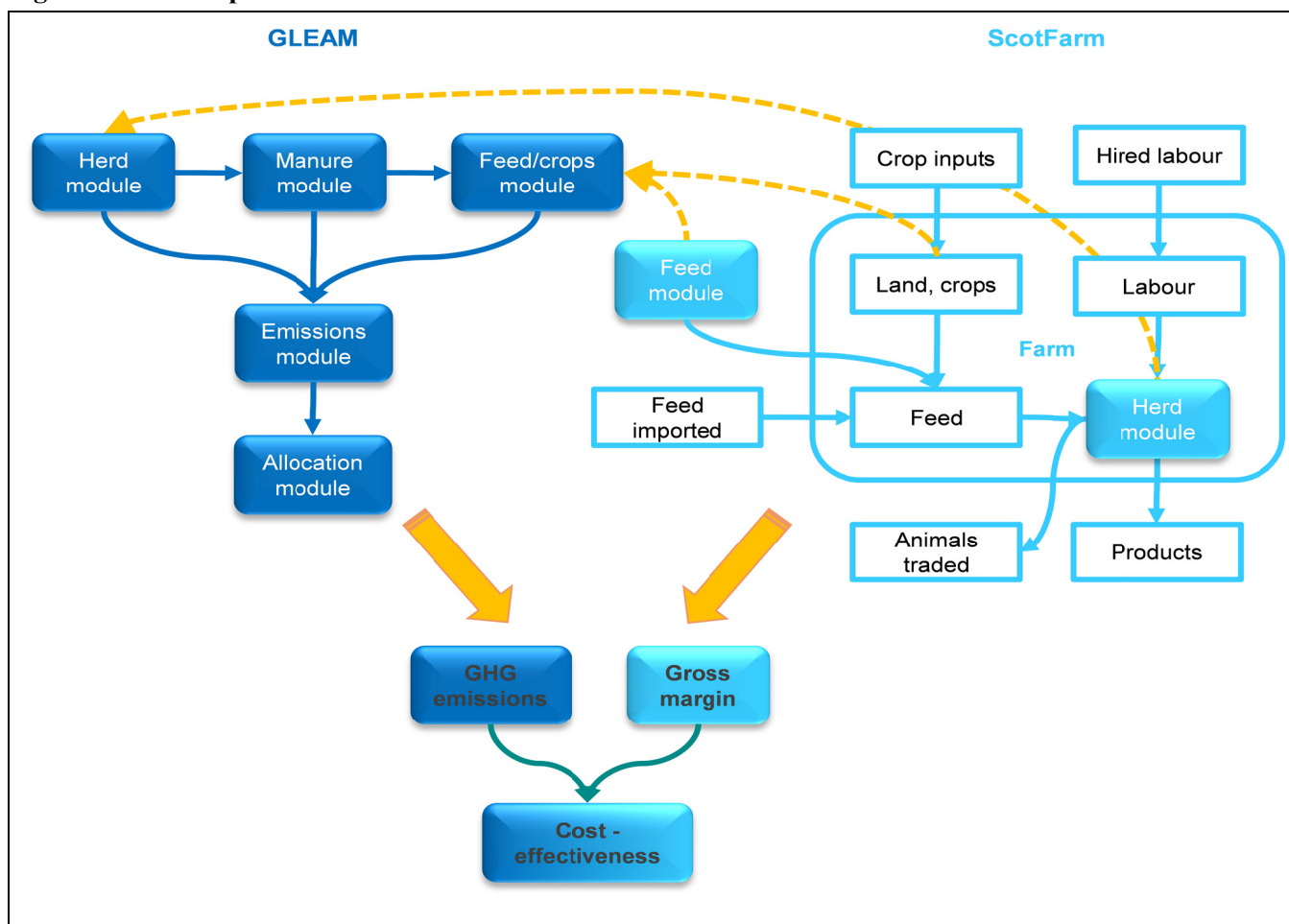
The model consists of all the major crops in Scotland. The initial land under these crops in each farm is

based on farm level data of the 2010/11 Farm Account Survey of Scotland (see section 2.4); however, the model allows land to be reallocated between these crops as well as transferred to grass production. The stocking rate on each farm is also fixed to the farm level data assuming that all farms were operating under optimum stocking rate. The dairy system has a four year replacement structure where dairy animals are culled after every four years. Similarly beef and sheep systems follow a two year replacement structure. The animals are replaced by on-farm or off-farm replacement stocks. A feed module, based on ALDERMAN and COTTRILL (1993) is used in the model to determine feed requirements for each of the animals on a farm based on type, age and production level of the animal. Feeds available to the livestock on farm are fresh grass, grass silage, grass hay, maize silage, grains, straw, beet and concentrate feeds.

### 2.3 Harmonising GLEAM and ScotFarm

Model parameters, input variables and modules are harmonised in GLEAM and ScotFarm in order to

**Figure 1. Conceptual framework of the two models**



Source: authors



**Table 2. Modelling differences between GLEAM and ScotFarm**

	<b>GLEAM</b>	<b>ScotFarm</b>
<b>Type of model</b>	Static, deterministic calculation over 1 year	Linear programming pseudo dynamic optimisation model with yearly time-steps
<b>System boundaries</b>	Partial LCA: GHG emissions from cradle-to-delivery at retail point	Farm gate
<b>Data input</b>	Primary data such as animal numbers, herd/flock parameters, mineral fertilizer application rates, temperature, etc. derived sources such as literature, databases and surveys (see MACLEOD et al., 2013, Appendix B).	Farm level data such as land area, land use, animal numbers and labour use; and financial data such as gross margins, variable costs and overhead costs are taken from Farm Account Survey (SCOTTISH GOVERNMENT, 2011). Farm coefficients such livestock units and labour requirements are taken from The Farm Management Handbook (SAC, 2012).
<b>Output</b>	Total annual commodity production (meat, milk and eggs); total GHG emissions; EI of each commodity.	Farm margins, feed rationing, herd size, land use; Total annual commodity production
<b>Dairy herd structure</b>	Six animal categories based on reproductive use and sex, herd structure is calculated using herd parameters	Four animal categories based on age and sex; herd structure is optimised based on herd parameters and prices
<b>Ration</b>	Imported from ScotFarm	Endogenous – the financially optimal combination of feed materials that can meet nutritional constraints is determined. The nutritional constraints are the metabolisable energy and protein requirements based on age and production level of individual animals (ALDERMAN and COTTRILL, 1993). Each of the farm groups however has to use concentrate diet at least 50% of level available in farm level data.

Source: authors

simulate the model farms and the mitigation measures' effect in parallel in both models. The herd structure, land use and feed ration composition are optimised in ScotFarm, and then exported to GLEAM (see Figure 1).

The main conceptual differences between the models are summarised in Table 2. To simulate both the baseline farms and the mitigation options in parallel in an optimisation and a static model, constraints are built in ScotFarm so that the farm structure of the baseline farm and the farm with the mitigation measure (apart from the specific changes due to the measure) is similar (i.e. the differences in grassland and arable land areas, herd size and feed composition between the farms modelled in GLEAM and in ScotFarm are not more than 5%). First the baseline farms are simulated in ScotFarm, and the resulting optimised baseline farm characteristics (land areas, number of cows, composition of the feed rations) are fed into GLEAM along with harmonised values for input parameters common to both models (e.g. milk and crop yields). The total production (of meat and milk) and GHG emissions are calculated in GLEAM and the farm gross margin is calculated in ScotFarm (see Figure 1). The procedure is then repeated for the scenario with the mitigation measure. The changes in emissions and in the EI of products due to the mitigation measure are then calculated by comparing the

results of the baseline scenario and the scenario with the measure.

## 2.4 Defining Farm Types

Farm level data was drawn from the 2010/11 Farm Account Survey of Scotland (SCOTTISH GOVERNMENT, 2011). The FAS consisted of farm level data from 484 farms which included physical as well as financial information of each of the sampled farms. A cluster analysis was carried out in SPSS<sup>1</sup> to group farms together with similar characteristics. Seven farm variables (production system, farm gross margins, land, animal number, labour, feed and milk yield) were used to group the farms. These variables were assumed to be the main differences between farms. The Squared Euclidean Distance Method was used in finding similarities between the farms. This method is commonly used in cluster analysis when there are multi-dimensional variables such as farm variables used in this study (SOLANO et al., 2001).

The cluster analysis resulted in fifteen farm types, with their main characteristics presented in (Table 3). These characteristics formed the basis of more detailed farm descriptions, which were generated

<sup>1</sup> SPSS is a statistical software. More details are available @ <http://www-01.ibm.com/software/analytics/spss/>

**Table 3. Typology of Scottish farms generated, based on FAS**

Farm types	Grass land (ha)	Arable land (ha)	Livestock units <sup>a</sup> (lu)	Variable costs (€ lu <sup>-1</sup> )	Labour <sup>b</sup> (man unit)
Dairy large	227.9	0.0	284	229.4	2.3
Dairy medium	99.5	11.7	137	227.7	2.1
Beef large	234.3	15.7	222	138.1	1.7
Beef medium	139.3	8.3	166	153.4	2.0
Beef small	77.0	4.5	84	143.0	1.3
Beef/Sheep large	263.5	27.9	242	151.2	2.9
Beef/Sheep medium	93.1	4.7	106	150.5	1.6
Sheep large	126.3	0.0	171	141.4	2.1
Sheep medium	65.3	0.0	81	126.0	1.5
Crop large	178.3	229.1	7	1428.6	7.5
Crop medium	86.3	218.0	8	1151.4	2.7
Crop small	46.6	89.0	3	1177.0	1.5
Mixed large	145.1	92.1	162	116.5	2.1
Mixed small	70.0	44.0	2045	112.5	1.6
Low land Beef/Sheep	172.0	9.0	162	124.3	1.8

<sup>a</sup> Livestock unit: (defined in terms of feed requirement) one unit equals to the maintenance of a mature 625 kg Friesian cow and the production of a 40-45 kg calf and 4,500 l of milk per year.

<sup>b</sup> man unit: 2,200 working hours year<sup>-1</sup>

Source: authors

to describe the baseline farms in terms of their cropping and livestock activities, fertiliser and feed use, crops and livestock product yields.

## 2.5 Case Study: Using Sexed Semen to Reduce Unwanted Male Calf Numbers on Scottish Dairy Farms

In Scottish dairy herds, a proportion of the cows are mated, usually by artificial insemination, using dairy breed semen to produce replacement stock, and the remainder are inseminated with beef semen to provide dairy x beef calves that are reared for beef production. The use of unsexed semen leads to significant number of pure dairy male calves, most of which are not required for replacement, and may be uneconomic to rear as beef animals (ROBERTS et al., 2008). This raises issues of economic and resource inefficiency and animal welfare. The use of sexed semen enables the number of cows mated with dairy semen to be reduced and the number of dairy x beef calves to be increased (see Table 4). The effect of using sexed semen on the emissions arising from dairy production and on the farm finances were investigated.

Representing common practice in Scotland, the baseline farms were assumed to use artificial insemination, using dairy semen on 70% of their cows and

heifers to produce enough female dairy calves for replacement (and as a 'by-product' dairy male calves, which are culled as newborns), and using beef semen on the remaining females to produce crossbred calves to be sold for rearing. With using sexed dairy semen the proportion of females inseminated with dairy semen is reduced to 40%, increasing the high-value crossbred calves proportion to 60%. The mitigation measure changes the income from the calves sold and the cost of the insemination in the financial model, and has effects on the GHG emissions from the reared beef cattle and on the meat produced.

**Table 4. Difference between the baseline farms and the farms with the mitigation measure implemented**

Variable	Baseline: unsexed semen	With sexed semen
Proportion of female dairy replacement calves	0.35	0.35
Proportion of male dairy calves	0.35	0.05
Proportion of crossbred calves	0.30	0.60
Increase in the variable cost due to using sexed semen (€ lu <sup>-1</sup> )	-	11.7

Source: authors

The sexed semen mitigation method is only applicable on farms with dairy cattle: i.e. dairy and mixed farms, but it is less relevant to mixed farms due to the much lower number of dairy cattle there, therefore only the medium and large dairy farms were modelled in this case study. The main farm characteristics are presented in Table 5.

Two important parameters in the financial and EI reduction performance of the mitigation measure are the additional cost of using the sexed dairy semen and the assumption on the EI of the suckler beef the additional crossbred calves are replacing. Sensitivity analysis was undertaken to explore the influence of these assumptions on cost-effectiveness.

**Table 5. Main characteristics of the modelled baseline**

Variable	Medium dairy farm	Large dairy farm
System: Year round calving, pasture based summer grazing for eight months, winter housing with grass silage feed, feed supplemented with concentrates and minerals year round.		
Number of cows (head)	149	300
Arable land area (ha)	11	0
Permanent grassland area (ha)	100	228
Milk sold (kg head <sup>-1</sup> year <sup>-1</sup> )	6,000	7,000
Milk price (€ l <sup>-1</sup> )	0.27	0.28
Crossbred calves' price (€ head <sup>-1</sup> )	100	86
Cow weight (kg head <sup>-1</sup> )	540	
Fertility rate of cows	0.87	
Fertility rate of heifers	0.95	
Calving period	all year	
Calving interval (month)	12	
Age at first calving (month)	28	
Replacement rate	0.25	
Milk wastage ratio ((milk secreted – milk sold) / milk secreted)	0.09	
Suckler beef EI (kg CO <sub>2</sub> e (carcass weight) <sup>-1</sup> )	30	

Source: authors

### 3 Results

Production, GHG emission and gross margin data of the baseline farms and the effect of using sexed semen are shown in Table 6. Producing more crossbred calves by using sexed semen increased the meat production of the systems by 47% for both medium and large dairy farms, while having no effect on milk production. This leads to an increase in the EI of the total protein produced, as a greater proportion of the protein is meat, which has a higher EI than milk. However, simply comparing the farms with and without sexed semen in term of the EI per unit of protein is misleading, as they are producing milk and meat in different proportions. In order to compare like with like, systems expansion can be used to isolate the emissions attributable to milk only. This is done by calculating the emissions that are avoided by producing beef, and subtracting these from the total emissions, to leave the emissions attributable to milk. In this

**Table 6. Production, GHG emission and gross margin data of the dairy farms with and without sexed semen**

		Medium dairy farm		Large dairy farm	
		Baseline	With SS <sup>a</sup>	Baseline	With SS
Production (kg protein year <sup>-1</sup> )	Meat	3,315	4,878	6,675	9,822
	Milk	29,591	29,591	68,815	68,815
GHG emissions for milk and meat (kg CO <sub>2</sub> e year <sup>-1</sup> )		2,144,750	2,366,120	4,559,644	5,005,356
EI of milk and meat protein (kg CO <sub>2</sub> e (kg protein) <sup>-1</sup> )		65.2	68.6	60.4	63.7
GHG emissions for milk only (kg CO <sub>2</sub> e year <sup>-1</sup> )		1,408,063	1,282,078	3,026,939	2,677,212
Milk EI (kg CO <sub>2</sub> e (kg milk) <sup>-1</sup> )		1.58	1.43	1.46	1.29
Gross margin (€ year <sup>-1</sup> )		165,284	167,128	261,569	264,120
Effect of mitigation measure		Medium dairy farm		Large dairy farm	
Change in milk GHG with SS (kg CO <sub>2</sub> e year <sup>-1</sup> )		-125,984		-349,727	
Change in gross margin with SS (€ year <sup>-1</sup> )		1,844		2,552	
Cost-effectiveness of SS (€ (t CO <sub>2</sub> e) <sup>-1</sup> )		-14.64		-7.30	

<sup>a</sup>SS: sexed semen

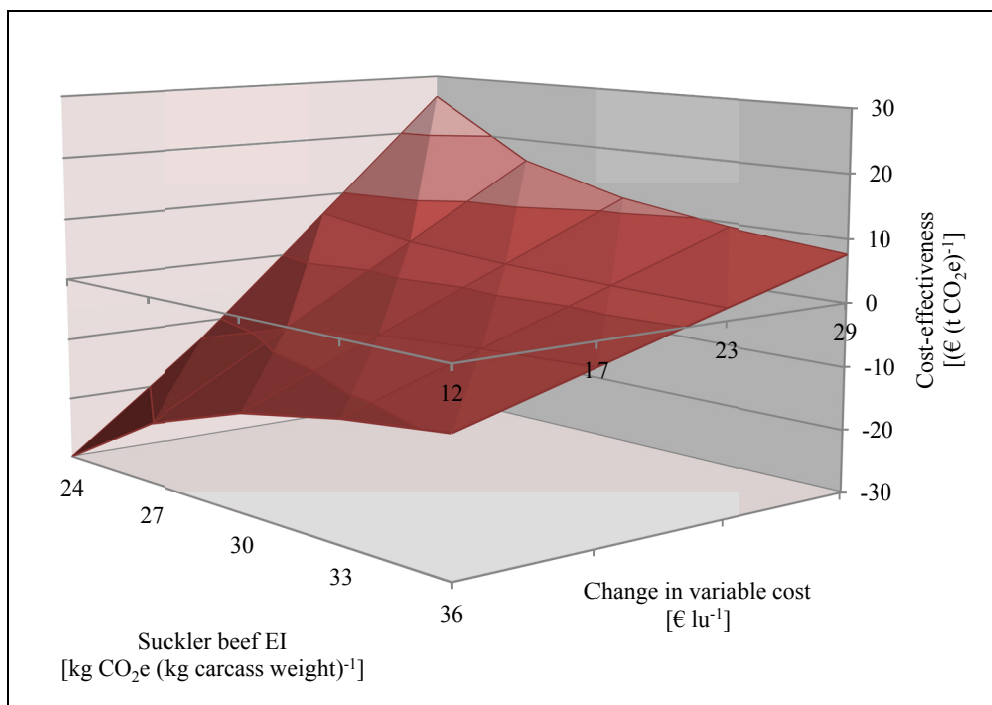
Source: authors



example it is assumed that if the beef was not produced by the surplus dairy calves, it would have to be produced by specialised (i.e. cow-calf) beef production. This type of beef production typically has significantly higher EI than that of dairy beef (see OPIO et al., 2013). It was assumed that the avoided specialised beef had an EI of 30 kg CO<sub>2</sub>e (kg carcass weight)<sup>-1</sup>. Under these assumptions, the EI of the milk is reduced by the mitigation measure by 9% and 12% on medium and large dairy farms, respectively. The financial modelling shows that the additional income from the increased number of marketable calves is more than 2.5 times more than the cost of sexed semen administration on both of the dairy farms. Therefore the cost-effectiveness of the measure on medium and large dairy farms is -15 and -7 € (t CO<sub>2</sub>e)<sup>-1</sup>, respectively.

The sensitivity analysis shows that varying the EI of the suckler beef by +20% and -20% changes the abatement potential by +55% and -55%, respectively, while changing the variable cost (due to sexed semen administration) by +50% reduces the net savings by 60% and increasing it by 100% or more makes a loss to the farmer. Overall, the cost-effectiveness of the measure varies between -33 and +27 € (t CO<sub>2</sub>e)<sup>-1</sup> (Figure 2).

**Figure 2. Sensitivity of the cost-effectiveness to the price of the sexed semen and the EI of the suckler beef**



Source: authors

## 4 Discussion

Developing more efficient agri-environmental policies requires the cost-effectiveness of GHG mitigation measures on different farm types to be quantified. The modelling framework proposed in this study provides this capacity, by using a financial optimisation model to simulate the gross margin changes and an LCA GHG model to estimate the emission changes arising from the mitigation measures. Adopting an LCA-approach in these calculations helps to ensure that mitigation measures do not simply displace emissions to other parts of the supply chain (although the danger of displacing production and emissions to other regions of the world still remains).

The current case study presents Scottish dairy farms as an example; however, both GLEAM and ScotFarm have the flexibility to model a wide variety of farm types and locations, provided input data of the requisite type and quality is available. Further benefits of the framework are the consistency in assumptions across mitigation measures and farm types and the inclusion of LCA and economic aspects to the whole farm modelling.

The modelling framework also has its limitations. The IPCC (2006) Tier 2 approach (PAUSTIAN et al., 2006) to livestock and manure emissions used in GLEAM provides considerable scope for varying livestock parameters and, in doing so, the modelling of mitigation measures. However, the Tier 1 approach to crop/soil emissions provides less scope (for example changes in the timing of fertiliser application or differences between soil types cannot be captured directly) and will be refined in the future versions of the model. The same applies to ScotFarm, where the cost breakdown distinguishes between labour, variable costs and overhead

costs, therefore the mitigation measures have to be described according to their effects on these variables rather than on more detailed farm activities. Nevertheless, these features also provide flexibility, as data collection at this level is quicker and often easier than acquiring farm type specific detailed activity and financial data. Therefore, the results should be interpreted as for the 'typical' farm in the modelled region rather than specific to one individual farm. It is also important to mention that the current framework does not capture the co-effects of GHG mitigation on other pollutants. These effects – especially on other types of reactive N (e.g. ammonia and nitrate) – can be significant for some mitigation measures, gaining even higher importance in regions with high nitrogen pollution. Nevertheless, these linked models provide a flexible and consistent way of calculating mitigation cost-effectiveness in a range of farm systems, helping to design better targeted regional and national policies for agriculture.

The results of the case study example show that using sexed semen on dairy farms might be a cost-effective way (i.e. cheaper than the shadow price of carbon), in some circumstances even win-win opportunity (i.e. providing financial savings to the farmers) to reduce emissions from cattle production. An important aspect of this GHG mitigation is that the GHG savings do not occur directly on the farm. High-yielding, specialised dairy and beef systems are inter-linked via the surplus calves in the dairy herds which can potentially be reared for meat and also via beef cross females from dairy herd becoming suckler cows. In the case of using sexed semen, the EI of the whole cattle system improves by decreasing the number of unwanted dairy male calves and increasing the amount of lower EI 'dairy beef' produced. The sensitivity analysis show that the measure stops generating financial savings on the farm after the additional cost of administering sexed semen exceeds approximately 21 €  $lu^{-1}$ . Similarly, the GHG savings are highly sensitive to the assumption on the emission intensity of the suckler beef production in the cattle system. The overall cost of sexed semen administration for the farmer depends not only on the cost of the semen but also on a number of factors related to fertility and herd management, like conception rate differences between cows and heifers, the availability of skilled personnel for the fertilisation, and the availability of sexed semen from high genetic merit sires. Providing more information and support in these areas to farmers would therefore increase the likelihood of the farmers achieving financial savings by

using sexed semen in dairy herds. All in all, the feasibility of integrating sexed semen use into the Scottish Government's GHG mitigation policy should be investigated.

## References

- ALDERMAN, G. and B.R. COTTRILL (1993): Energy and protein requirements of ruminants. An advisory manual prepared by the AFRC Technical Committee on responses to nutrients. CAB International, Wallingford, UK.
- ALEXANDRATOS, N. and J. BRUINSMA (2012): World agriculture towards 2030/2050: the 2012 revision. ESA Working paper No. 12-03. FAO, Rome.
- BELLARBY, J., R. TIRADO, A. LEIP, F. WEISS, J.P. LESSCHEN and P. SMITH (2013): Livestock greenhouse gas emissions and mitigation potential in Europe. In: *Global Change Biology* 19 (1): 3-18.
- COTTLE, D.J., J.V. NOLAN and S.G. WIEDEMANN (2011): Ruminant enteric methane mitigation: a review. In: *Animal Production Science* 51 (6): 491-514.
- CROSSON, P., L. SHALLOO, D. O'BRIEN, G.J. LANIGAN, P.A. FOLEY, T.M. BOLAND and D.A. KENNY (2011): A review of whole farm systems models of greenhouse gas emissions from beef and dairy cattle production systems. In: *Animal Feed Science and Technology* 166-167: 29-45.
- FORESIGHT (2011): The Future of Food and Farming: Challenges and choices for global sustainability. The Government Office for Science, London.
- GERBER, P., T. VELLINGA, K. DIETZE, A. FALCUCCI, G. GIANNI, J. MOUNSEY, L. MAIORANO, C. OPIO, D. SIROINI, O. THIEME and V. WEILER (2010): Greenhouse Gas Emissions from the Dairy Sector - A Life Cycle Assessment. FAO, Rome.
- HENNESSY, T., S. SHRESTHA and M. FARRELL (2008): Quantifying the viability of farming in Ireland: can decoupling address the regional imbalances? In: *Irish Geography* 41 (1): 29-47.
- HRISTOV, A., J. OH, C. LEE, R. MEINEN, F. MONTES, T. OTT, J. FIRKINS, A. ROTZ, C. DELL, A. ADESOGAN, W.Z. YANG, J. TRICARICO, E. KEBREAB, G.C. WAGHORN, J. DIJKSTRA and S. OOSTING (2013): Mitigation of greenhouse gas emissions in livestock production – A review of technical options for non-CO<sub>2</sub> emissions. In: Gerber, P., B. Henderson and H. Makkar (eds.): *FAO Animal Production and Health Paper 177*. FAO, Rome.
- IPCC (Intergovernmental Panel on Climate Change) (2006): 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Prepared by the National Greenhouse Gas Inventories Programme. In: Eggleston, H.S., L. Buendia, K. Miwa, T. Ngara and K. Tanabe (eds.). Institute for Global Environmental Strategies (IGES), Japan.
- (2013): Summary for Policymakers. In: T.F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.): *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate*

- Change, WGI AR5 PM. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
- LOUHICHI, K., A. KANELLOPOULOS, S. JANSSEN, G. FLICHTMAN, M. BLANCO, H. HENGSDIJK, T. HECKELEI, P. BERENTSEN, A.O. LANSINK and M.V. ITTERSUM (2010): FSSIM, a bio-economic farm model for simulating the response of EU farming systems to agricultural and environmental policies. In: *Agricultural Systems* 103 (8): 585-597.
- MACLEOD, M., P. GERBER, A. MOTTET, G. TEMPIO, A. FALCUCCI, C. OPIO, T. VELLINGA, B. HENDERSON and H. STEINFELD (2013): Greenhouse gas emissions from pig and chicken supply chains – A global life cycle assessment. FAO, Rome.
- NEUFELDT, H. and M. SCHAFER (2008): Mitigation strategies for greenhouse gas emissions from agriculture using a regional economic-ecosystem model. In: *Agriculture Ecosystems & Environment* 123 (4): 305-316.
- OPIO, C., P. GERBER, A. MOTTET, A. FALCUCCI, G. TEMPIO, M. MACLEOD, T. VELLINGA, B. HENDERSON and H. STEINFELD (2013): Greenhouse gas emissions from ruminant supply chains – A global life cycle assessment. FAO, Rome.
- ROBERTS, D.J., J. BELL, K. BEVAN and T. McEVOY (2008): Beyond calf exports: the efficacy, economics and practicalities of sexed semen as a welfare-friendly herd replacement tool in the dairy industry. Report produced for Compassion in World Farming & The Royal Society for the Prevention of Cruelty to Animals. Scottish Agricultural College, Edinburgh.
- SAC (Scottish Agricultural College) (2012): The farm management handbook 2012/13. SAC Consulting, Edinburgh.
- SCHILS, R.L.M., J.E. OLESEN, A. DEL PRADO and J.F. SOUSANA (2007): A review of farm level modelling approaches for mitigating greenhouse gas emissions from ruminant livestock systems. In: *Livestock Science* 112 (3): 240-251.
- SCOTTISH GOVERNMENT (2011): Economic report on Scottish Agriculture. Rural and Environment Science and Analytical Services (RESAS), Scottish Government. In: <http://www.scotland.gov.uk/Publications/2011/06/15143401/0>.
- SHRESTHA, S. (2004): Adaptation strategies for dairy farms in central and north-west England under climate change. PhD thesis. University of Nottingham.
- SHRESTHA, S. and T. HENNESSY (2006): Changes in land use on farms in the Border and the Mid-East regions of Ireland after decoupling of farm payments. *Agricultural Research Forum*, 15-16 March, Tullamore, Ireland.
- SHRESTHA, S., T. HENNESSY and S. HYNES (2007): The effect of decoupling on farming in Ireland: a regional analysis. In: *Irish Journal of Agriculture and Food Research* 46 (1): 1-13.
- SHRESTHA, S., P. CIAIAN, M. HIMICS and B. VAN DOORSLAER (2013): Impacts of climate change on EU agriculture. In: *Review of Agricultural and Applied Economics* 16 (2): 24-39.
- SOLANO, C., H. LEON, E. PEREZ and M. HERRERO (2001): Characterising objectives profiles of Costa Rican dairy farmers. In: *Agricultural Systems* 67 (3): 153-179.

## Acknowledgements

This analysis has been undertaken with Scottish Government Rural and Environmental Science and Analytical Services division (RESAS) funding to SAC and to ClimateXChange, and as part of the Animal-Change project which received funding from the European Community's Seventh Framework Programme (FP7/ 2007-2013) under the grant agreement n° 266018.

The authors are grateful to the UN FAO for permission to use GLEAM in this study.

Contact author:

**VERA EORY**

Research Division, SRUC, West Mains Road  
Edinburgh, EH9 3JG, UK  
e-mail: vera.eory@sruc.ac.uk